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Fuel Dynamics in Southern Pine Beetle-Killed Stands and Their Implication to Fire Behavior

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FUEL DYNAMICS IN SOUTHERN PINE BEETLE-KILLED STANDS AND THEIR IMPLICATION TO FIRE BEHAVIOR

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ABSTRACT

The increase in both southern pine beetle (SPB) (*Dendroctonus frontalis* Zimm) outbreak and wildfire occurrence in recent decades has resulted in a growing concern regarding possible interactions. Few studies have quantified fuel characteristics of SPB-killed stands, and none has studied the dynamics of this fuel complex over time. Moreover, how changes in these fuels affect fire behavior remains unknown. To address this lack of empirical data, field measurements and modeling were combined to study fuel dynamics and potential fire behavior within control and post-outbreak loblolly pine-dominated forest stands across a chronosequence ranging from 0 to 8 years since outbreak. Fuels data were collected on three study areas within the Piedmont of Georgia and South Carolina, USA. Fuel loading was significantly greater in post-outbreak stands than in control stands for several types of fuels. Stand structure was altered between stand types, containing fewer live pines and more hardwoods in post-outbreak stands. The Fuel Characteristic Classification System (FCCS) was used to construct representative fuelbeds from the measured data. The customized fuelbeds were then used to model surface fire behavior as a measure of the change in fuel loading and stand structure resulting from SPB outbreak. Flame length and rate of spread (ROS) were derived as metrics of predicted surface fire behavior. BehavePlus was used to further understand the consequences of these fuel changes. Both FCCS and BehavePlus predicted faster ROS and higher flame lengths in post-outbreak stands than in uninfested stands. Stands measured 8 years after outbreak under extremely dry moisture conditions were predicted to have the most extreme fire behavior. These results were compared to existing standard fuel models and discussed in the context of management implications for SPB-killed Piedmont forests.

DEDICATION

I dedicate this thesis to my parents and grandparents, all of whom have instilled within me a passion for life and learning.

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CHAPTER ONE

INTRODUCTION

Bark beetles in the genus *Dendroctonus* (Coleoptera: Curculionidae, Scolytinae) are native insects that play an important role in North American forests. At low population densities during non-outbreak years, bark beetles serve as a natural means of mortality for stressed pines, being most often associated with senescent or lightning-damaged trees (Coulson *et al.* 1983; Xi *et al.* 2010). By shifting old and injured trees from the canopy to the forest floor, bark beetles inadvertently create wildlife habitat, produce openings for regeneration, increase fuel loading, and contribute to nutrient cycling (USDA Forest Service 2005). However, forest ecosystem function and sustainability may be negatively affected when bark beetle populations reach epidemic densities. By acting as invasive agents, they can pose serious threats to forest ecosystem health.

Local outbreaks start in a single or a few host trees then spread to nearby trees, creating a "spot." Under optimal host and environmental conditions, as when a large number of susceptible host trees have been subjected to a period of stress, eruptive outbreaks can occur. Forest stands that have been damaged by storms, fire, logging, or drought are more at risk for attack than unstressed stands (Price 1994). Land use changes over the past century including fire exclusion, harvesting, short-rotation forestry, and conversion of natural stands to high-density pine plantations (Cameron and Billings 1988; Clarke *et al.* 2000; Schowalter *et al.* 1981) as well as erosion and decreased soil fertility (Karpinski *et al.* 1997) have promoted the establishment of stressed forests.

Aggressive bark beetle species, such as the southern pine beetle (SPB) (*Dendroctonus frontalis* Zimmermann), are able to overcome host tree resistance in these stressed stands. This ability has made SPB the most serious ecologic and economic pest of pine forests throughout the southeast U.S.

Periodic bark beetle outbreaks have regularly occurred within coniferous forests for at least 13,000 years (Jenkins 2011). However, in recent decades SPB outbreaks have been severe and persistent in several regions, including the Piedmont physiographic providence (Ward and Mistretta 2002). The most damaging SPB outbreak in recent years lasted from 1999 to 2003, impacting almost one million acres of forest land across eight states, and causing an estimated $1.5 billion in losses (USDA Forest Service 2005*a*). In South Carolina alone, SPB infestations affected over 13.5 million hectares from 2000 through 2002 (USDA Forest Service 2003) and caused over $250 million in damages for 2002 (USDA Forest Service 2002). Another large-scale outbreak, albeit less severe than the previous outbreak, occurred across several Piedmont states from 2007-2008. Continuous drought conditions combined with erosion and a depleted soil resource due to the Piedmont's long legacy of crop agriculture, livestock pasturing, and exploitative timber harvesting (Callaham *et al.* 2006; Stottlemyer 2011) are the most likely factors to blame for these recent increases in outbreak severity.

The increase in both SPB outbreak and wildfire occurrence in recent decades has resulted in a growing concern regarding possible interactions. The potential for even greater devastation in the future due to increasing drought conditions and a warming climate (Gan 2004; Jenkins 2011; Waring *et al.* 2009) makes research regarding their relationship more urgent than ever. Numerous studies have examined the effects of wildland fire on bark beetle activity (Hanula *et al.* 1999; McHugh *et al.* 2003; Rasmussen *et al.* 1996; Santoro *et al.* 2001), allowing us to understand the factors associated with wildfire that can lead to beetle outbreak. However, in comparison, very few reverse studies have been conducted on the consequences of bark beetle outbreak on subsequent wildfire effects and behavior.

A majority of the research that has been conducted on bark beetle-driven influence on fuel loading and potential wildfire behavior has been focused on mountain pine beetle (MPB) (*D. ponderosae* Hopkins) outbreak in lodgepole pine (*Pinus contorta* Dougl. ex Loud) forests (Brown 1975; Geiszler *et al.* 1980; Lotan *et al*. 1985; Lynch *et al.* 2006; Page and Jenkins 2007*b*; Romme *et al.* 1986*b*; Schoennagel *et al.* 2012; Simard 2010; Simard *et al.* 2011). Fewer studies have examined the influence of spruce beetle (*D. rufipennis* Kirby) outbreak on subsequent fire extent and severity within Colorado sub-alpine forests (Bebi *et al.* 2003; Bigler *et al.* 2005; Kulakowski *et al.* 2003; Kulakowski and Veblin 2007). Jenkins *et al.* (2008) described the changes to fuel bed characteristics and predicted fire behavior resulting from Douglas-fir beetle (*D. pseudotsugae* Hopkins), spruce beetle, and MPB activity on Intermountain conifer forests. Little research has been conducted to describe the relationship between bark beetles and their influence on wildfire in other forest systems, particularly for pine forests within the Piedmont of the eastern U.S.

Severe SPB outbreaks can produce up to 100 percent mortality of pines over an area of a hectare or more. As a result, fuel loading in these SPB-killed spots increases suddenly and dramatically soon after outbreak. As pines fall to the ground, piled logs, dense under-brush, and conditions which are dangerous and unsightly are created (Stottlemyer 2011). It has often been presumed that the high loads of surface and canopy dead fuels created by SPB outbreak may increase wildfire susceptibility and/or increase the probability of extreme wildfire behavior (thereby also increasing the difficulty of fire control), however, empirical data to support this presumption are limited, and definitive results are lacking.

A few studies have been conducted to quantify fuel characteristics of SPB-killed stands (Stottlemyer 2011; Waldrop *et al.* 2007), but none have studied the dynamics of this fuel complex. Currently, no studies that have quantified potential or observed wildfire behavior in SPB-killed forests are available. Considering the importance of both bark beetles and fire in the ecology and management of southeastern forests, this is a serious concern. Therefore, the goal of this project was to study fuel dynamics and their implication to fire behavior in loblolly pine (*Pinus taeda* L.) dominated forest stands killed by SPB outbreak within the Piedmont ecoregion. Specifically, the objectives were to (1) provide baseline quantification of live and dead fuels in post-outbreak stands killed by SPB at different years, (2) characterize and compare fuel dynamics of unaffected stands with those of SPB-killed stands, and (3) model predicted fire behavior by using measured fuel data to understand the consequences of these fuel changes.

CHAPTER TWO

LITERATURE REVIEW

This literature review will describe the current understanding of the influence of southern pine beetle (SPB) on fuels and wildfire behavior in loblolly pine-dominated forest ecosystems within the Piedmont ecoregion. In addition, detail will be provided in regards to the fire behavior modeling programs used for this study.

Southern Pine Beetle within the Piedmont

Southern pine beetles are indigenous to North and Central America. Their distribution ranges from New Jersey to Florida and west to Texas, and from Arizona and New Mexico to Nicaragua. This range encompasses the entire Piedmont ecoregion. Climatic changes that occur throughout these regions influence SPB directly, through physiological processes, and indirectly, through its host trees and natural predators (Gan 2004). Climate, especially temperature, plays a large role in most aspects of SPB biology. Temperature can directly affect their survival, with their northern range limited by lethal winter temperatures (Ungerer *et al.* 1999). Like most insects, they have temperature-dependent developmental rates, oviposition rates, and re-emergence rates and must therefore be able to respond to annual climatic variation. Extreme or extended periods of high temperature during the summer (> 90°F) decrease the developmental rates of most SPB life stages and result in a greater tendency of outbreak spots to collapse (Friedenberg *et al.* 2008). More generations are produced per year when winters are short and warm (Thatcher and Pickard 1964; Ungerer *et al.* 1999; White and Franklin 1976).

Although not as influential as temperature, precipitation can indirectly affect SPB. When drought conditions occur, host trees become stressed and are less resistant to SPB attack. The most severe multi-year droughts in recent South Carolina history occurred from 1998-2002 (Gellici *et al.* 2004) and 2007-2008 (http://www.dnr.sc.gov/climate/sco/Drought/drought_press_release.php), synchronous with the most severe SPB outbreaks in recent South Carolina history. Increasing drought conditions and a warming climate may soon cause attacks to occur at an even greater scale (Gan 2004; Jenkins 2011; Waring *et al.* 2009). However, bark beetle outbreaks cannot occur without susceptible (typically overly-dense and unthinned) stands.

SPB exhibit cyclic populations that fluctuate from low endemic to high outbreak densities. During initial infestation, tree mortality is generally restricted to one or only a few individual stressed and dying pines per hectare. In the southern United States, preferred hosts include loblolly pine, shortleaf pine (*P. echinata* Mill.), pitch pine (*P. rigida* Mill.), pond pine (*P. serotina* Michx.), and Virginia pine (*P. virginiana* Mill.) (Clarke and Nowak 2009; Thatcher and Barry 1982). Shortleaf pine is among the most susceptible to attack within the Piedmont (Thatcher and Barry 1982). During the endemic phase, unhealthy, weakened host pines are typically attacked first, but as the beetle population increases to outbreak status, healthy pines or even those species which are generally considered to be more resistant to SPB attacks, including slash pine (*P. elliottii* Engelm.), longleaf pine (*P. palustris* Mill.), spruce (*Picea* spp.), hemlock (*Tsuga* spp.), and juniper (*Juniperus* spp.) may also be affected (Belanger *et al.* 1993; Clarke and Nowak 2009).

Outbreaks can occur when otherwise healthy, but susceptible stands, are subjected to a period of short-term stress from drought, fire, or other disturbance. Susceptible stands are typically dense stands comprised of a large percentage (>60%) of mature, large diameter host trees (Jenkins 2011). Stands containing basal areas greater than 100 ft^2/acre have a considerably higher risk of beetle attack and spot spread than those with lower densities (Coster and Searcy 1980). Changes to fuel complexes and fire behavior due to fire suppression and exclusion policies have created an abundance of these old, dense conifer stands within the Piedmont.

Over 60 percent of all SPB infestations within the Piedmont occur on slopes greater than 10 percent, possibly due to the stress on the trees from erosion (Karpinski *et al.* 1997). In addition, Piedmont pine stands growing on soils with a clay content of at least 28 percent are more susceptible to attack (Karpinski *et al.* 1997). Once a tree is attacked, spots (i.e., infestations of multiple trees) may expand at rates of up to 50 feet per day (Meeker *et al.* 2008). An analysis of about 26,000 spots from National Forests in 11 states of the southeastern U.S. revealed that the median initial size of the spots was 14 trees, with a majority (90%) ranging between four to 50 trees (Friedenberg *et al.* 2007). SPB outbreaks typically occur within the Piedmont every 7 to 10 years (Price *et al.* 1998) for a duration of approximately 2 to 4 years (Hedden 1978), ending when most large diameter trees have been killed and SPB population levels decline. Stands then enter the post-outbreak phase, which lasts for decades to centuries until small surviving or newly regenerated host trees reach susceptible age and size once more.

A successful SPB attack on a host tree follows a well-studied sequence. First, adult beetles initially disperse by flight, probably at random. When a suitable host is found (i.e., an old, weakened, or moribund pine), beetles immediately begin feeding and nesting activities in the bark. Once a tree has been colonized, it rarely survives. Winding S-shaped feeding and nesting galleries are constructed in the cambium, effectively girdling the tree. The inner bark (phloem), which is rich in nutrients and is used by all life stages of the SPB, changes dramatically following colonization. The nutrient-poor outer bark is used sparingly, and has more stable conditions than the inner bark. Potent attractant pheromones are released into the air, luring other beetles to begin feeding and nesting within the same host tree, thus producing a tightly synchronized pheromone-induced mass attack (Clarke and Nowak 2009). As the larvae develop, emerge, and disperse, the cycle begins anew (Moeck *et al.* 1981; Raffa *et al.* 1993). Anti-aggregation pheromones are eventually released to discourage additional beetles from coming to the original host tree when it is nearing carrying capacity.

In addition to the girdling effect caused by the feeding and nesting galleries, most beetles carry a blue stain fungus (*Ophiostoma minus* Sydow) which contributes to the death of the tree by entering the trees' vascular system and plugging it, thus ultimately preventing the movement of nutrients. The degree of blue stain development is dependent upon degree of host colonization, fungal pathogenicity, host resistance and the ability of the tree to compartmentalize the fungi (Jenkins 2011). Therefore, the amount of sapwood that is affected varies considerably, but the fungi will regardless reduce water flow to the crown resulting in a net reduction in foliar moisture content (FMCn). As time goes on, the FMCn continues to decrease as blue stain development increases.

Trees attacked by SPB are typically divided into three categories: fresh attacks (green phase), faders (red phase), and vacated (gray phase). Fresh attacks are distinguished by the presence of adult SPB, clerid beetles (*Thanasimus dubius*), tight bark with white phloem, green crowns, and boring dust (Clarke and Nowak 2009). Once the larvae hatch and begin feeding, the foliage begins to change color (Clarke and Nowak 2009). The discoloration of needles in the tree crown is the most obvious symptom of an attack. Needles fade from green to dull green to yellow, and finally to reddish-brown before falling. Once a tree has been vacated, hardened pitch tubes, loose bark, and numerous exit holes are observable, the needles are either red or have fallen off, and adult clerid beetles are no longer present (Clarke and Nowak 2009).

Widespread mortality of host trees from SPB outbreak results in the death of millions of dollars worth of both pine timber and residential landscape pines. Loblolly pine is the leading commercial timber species in the southeastern United States (Baker and Langdon 1990) and is primarily used for lumber and pulpwood. Loblolly pine is a rapidly growing, medium- to large-sized native, evergreen conifer with an average height at maturity of 89-112 feet and a diameter of 24-30 inches (Carey 1992). Within the South Carolina Piedmont, the most commonly attacked stand types are mixed shortleaf pine-hardwood stands as well as dense, naturally-regenerated loblolly pine stands and unthinned loblolly pine plantations (Stottlemyer *et al.* 2012). Upland forests within the Piedmont are typically comprised of pine or mixed pine-hardwood. According to a

USDA Forest Service timberland survey, loblolly-shortleaf pine forests occupy over two million acres in South Carolina's Piedmont, followed by oak-hickory (Conner and Sheffield 2001). The widespread range of host trees within in the Piedmont allow the potential for continued future outbreak.

Wildfire within the Piedmont

Prior to European settlement, lightning was a dominant natural cause of wildfire, and lightning-caused fires (in combination with topography and climate) played a major role in shaping vegetative communities of the southeastern United States (Dixon 1989; Pyne 2001; Van Lear and Waldrop 1989). These fires typically began during summer months, and burned extensive acreages. In the Piedmont, loblolly pine and loblolly-shortleaf pine forests experienced surface fires that were frequent, low intensity, and sustained by fine grass and pine needles (Stanturf *et al.* 2002; Wade *et al.* 2000). According to Wade *et al.* (2000), loblolly pine stands were historically confined to wetter sites due to their susceptibility to fire when young, and had mean fire intervals of three to eight years in the pre-settlement era. These naturally occurring low-intensity fires minimized fuel accumulation and rarely became crown fires.

Historical records suggest that Native Americans used fire as one technique to provide better access, open the forests for hunting and farming, and encourage growth of desirable understory species for food and medicinal purposes (Dixon 1989; Van Lear and Waldrop 1989). Evidence shows that the arrival of European settlers in the early 19th century increased the number of fires; however, fire was later seen as a problem and effort was made to suppress every fire, regardless of the cause (Pyne 1997). Today, traditional management practices have disrupted the natural interactions between pine forests, SPB, and fire in the Piedmont. Fire suppression tactics are still regularly employed and mean fire intervals have been extended to more than 100 years (Knebel and Wentworth 2007). South Carolina alone suppresses approximately 5,000 to 6,000 wildfires each year (Mohr *et al.* 2004; Mohr and Waldrop 2006). These all-too-successful fire suppression efforts have led to fires that consume extensive areas (i.e., megafires) and have created dense forest stands which provide large fuel stores of woody debris that periodic fires used to eliminate. Absence of fire has led to forest stands dominated by species such as maple, beech, and sweetgum. In addition, loblolly pine forests have dramatically increased in acreage.

The arrival of humans as a source of ignition changed the pattern of wildfire regimes, and now fires in very few parts of the world are unaffected by human influence. Prior to the emergence of man in the southeastern region, the primary burning season was the late spring and summer months of May-August, with fires primarily ignited by lightning from thunderstorms (Gambrell 2008; Knapp *et al*. 2009). Today, a majority of wildfires within the South Carolina Piedmont are human caused, either by arson or accidentally from ignition sources such as campfires or debris burning (data obtained from SCDOF). In fact, a review of wildfire occurrence data in South Carolina from 1997-2009 found that lightning accounted for only 6% of the acreage burned (data obtained from SCDOF). Not surprisingly, a pattern of wildfire ignition points has evolved from human influence, with most fires tending to occur along roads and rivers (paths taken by smokers, campers, hikers, and other recreational enthusiasts) (Brown *et al*. 2002). In addition to human influence on wildfire, prescribed fire is a growing practice used by southern resource managers to achieve various forest management objectives ranging from fuel reduction to improved wildlife habitat. Although skills for prescribed burning in the region are progressing, fire managers lack access to basic fuel loading and fire behavior information that are readily available in other regions (Waldrop *et al.* 2007).

Fire behavior is primarily dependent on local topography, relative humidity, wind speed, temperature, and fuel moisture (Rothermel 1983; Scott and Burgan 2005). Fire behavior also differs from site to site, in part due to varying understory species assemblages, even in plantations. Available fuels provided by plants can intensify or lessen fire behavior, depending on the foliar chemistry of the species (Nowacki and Abrams 2008). Changes in fuel moisture can impact the probability of ignition as well as fire intensity and/or duration. Variable site and environmental conditions result in wildland fires that burn with irregular intensity and unpredictable severity (Turner *et al.* 1997). As a topographical and ecological transition zone between the Coastal Plain and Appalachian Mountains, the Piedmont contains both pine and hardwood species from these adjacent regions, oftentimes with both species occurring together in mixed pine-hardwood stands. Despite the variability in topography, vegetation, and other environmental conditions between forest stands, the fire regime remains fairly consistent throughout the Piedmont.

Fire regimes are largely influenced by regional climate patterns, which regulate fire occurrence (i.e., number of fires per unit time in a specified area), behavior, and spread (Pyne 2001). The fire season differs slightly from state to state within the Piedmont, but all occur during the winter or early spring months when most of the vegetation is dead or dormant. In addition to climate, stand and site conditions also assist in shaping the fire regime within the Piedmont, particularly the amount of pine versus hardwood in a stand (Stanturf 2008). The Piedmont contains a large amount of loblolly and shortleaf pines, which have an understory (surface) fire regime (Wade *et al.* 2000). Surface fires are those fires that burn litter and other live and dead fuels (grass, shrubs, dead and down limbs, needle and leaf litter, debris, etc.) at or near the ground surface, mostly by flaming combustion (Pyne 1996). All wildland fires, except those that smolder in snags or punky trees struck by lightning, begin as surface fires.

Studies of disease or insect disturbances linked with wildfire often highlight the importance of crown fire since these types of fires can quickly produce large, long-term ecological changes to the landscape (Valachovic *et al.* 2011). However, many of these studies occur in western ecosystems where crown fires are the dominant fire type. Although crown fires have been documented to occur within loblolly forest stands in the Piedmont (Oosting 1944), surface fires are the principal fire type within the region, and were therefore the focus for this study. Wildland fires have become important in the ecology and management of southeastern forests, but surface fire behavior within SPB-killed forests during these fires is currently unknown.

Southern Pine Beetle and Wildfire as Interacting Ecological Disturbances

Disturbance is a key component of ecological systems. Disturbances may be biotic (i.e., disease or insect pest), abiotic (i.e., hurricane or tornado), or a combination of the two (i.e., fires require a source of fuel (biotic) as well as conditions suitable for burning (abiotic)). The term "disturbance" has been defined in various ways, one of the most common of which was defined by White and Pickett (1985) as "any relatively discrete event that disrupts the structure of an ecosystem, community, or population, and changes resource availability or the physical environment." Although disturbances can, and will, interact with one another, these interactions can be complex and are poorly understood, resulting in a surprisingly low number of studies conducted on the topic.

Prior disturbance can be a strong determinant of ecosystem response to a subsequent disturbance (Everham and Brokaw 1996; Paine *et al.* 1998; Turner 2010), however, little information is available regarding whether and when a disturbance will intensify the effects of another or change the probability of its occurrence (Turner 2010). Natural disturbance events of various size and scale, including fires and SPB activity, are inherent components of southern pine forests and greatly influence forest structure, composition and dynamics in the southern United States (Stanturf *et al.* 2007). SPB and fire historically occurred in a balanced relationship that promoted the stability of coniferous forests throughout the southeastern U.S. (Schowalter *et al.* 1981) by serving as natural harvesters. Together, fire and SBP likely "maintained uneven-aged pine forests and successional openings on upland sites, as well as diversity of herbaceous, pine-hardwood, and hardwood lowland communities" (Coulson 1980). Periodic fires acted as an ecologically renewing force by restarting vegetation development within patches of the ecosystem, thus creating communities at various stages of succession. SPB may have aided this process by thinning old or stressed stands as a means of maintaining the health and diversity of the forest community, by opening the canopy to enhance the effect of wind, and by providing heavy fuel loads to enhance the effect of subsequent fire (Coulson 1980; Schowalter *et al.* 1981). As a result, the relative stability of the ecosystem increased.

Bark beetles are one of few native agents in nature with the ability to rapidly alter fuel complexes over large spatial scales. Jenkins *et al.* (2008) described the effect of Intermountain conifer forest altered fuel complex on the principle fire behavior descriptors including rate of spread, fireline intensity, and flame length over the course of a MPB rotation. They found that MPB epidemics caused a substantial change in species composition and a highly altered fuels complex. Early in the red phase (dead trees with red needles) they found a net increase in the amount of fine surface fuels when compared to endemic stands. Large, dead, woody fuels, and live surface fuels dominated the gray stage (trees with no needles).

Today, fire suppression efforts within the Piedmont have created dense forest stands which provide large fuel stores of woody debris for wildfires that periodic fires used to eliminate. In addition to creating heavy fuel loads, these unburned stands contain dense understory vegetation in need of increased nutrient demand. Eventually, light and nutrient demand may exceed availability, resulting in dense, stressed stands susceptible to SPB attack. Eventually, as pines die out due to lack of nutrients or by succumbing to SPB attack, the shade-tolerant hardwoods become established and can outcompete the pines for dominance.

Maine (1979) provided a rough estimate that approximately 64 acres of SPB spots burned annually in the southern U.S. at the time of his study. This estimate was based on 0.25892 percent of commercial forest acreage in the South being burned annually, 0.05 percent of the commercial forest containing SPB spots, and 49.4 million acres of loblolly-shortleaf pine. This estimate was based on existing numbers at the time of his study, assumed that outbreak characteristics within a single study site (i.e., Leuschner *et al.* 1976) held true southwide, and assumed fire incidence was independent of SPB attack. As both SPB attacks and wildland fire (prescribed fire and wildfire) have increased in occurrence and severity since Maine's earlier 1970's estimate, it is expected that overlap between fires and SPB spots have greatly increased since that time and will continue to increase in the foreseeable future. As more SPB-killed stands begin to burn in wildland fire situations, it will become increasingly necessary for land managers to have the ability to predict fire behavior in these areas.

Global mean surface temperatures in the year 2010 ranked among the top two warmest years since the start of surface instrumental record keeping in the late 19th century (Thorne 2011). Many forecasters believe that the southeast will continue to experience rising temperatures in the coming decades. Global climate change may alter disturbance regimes in the future since many disturbances are dependent on climate, including SPB outbreak. It is generally thought that increased MPB activity, and thus increased pine mortality rates, across millions of hectares of forests in the western U.S.

are primarily a consequence of broad-scale warming and are likely to continue with expected future warming (Bentz *et al.* 2009, Hicke *et al.* 2006, Romme *et al.* 2006). SPB are killed by deep-freeze winters, but may be able to persist much longer through the winter as temperatures warm. Westerling *et al.* (2006) demonstrated a strong link between climate change and increased wildfires. The need to understand the SPB-wildfire interaction is urgent as both SPB outbreak and wildfire occurrence begin to increase.

SPB-Altered Fuels on Potential Fire Behavior and Effects

The down dead fuels that result from SPB attack substantially influence fire behavior and fire effects. Fuels are often defined as the physical characteristics (i.e., loading, depth, height, and bulk density) of live and dead biomass that contribute to wildfire (Davis 1959, Riccardi *et al.* 2007). Smaller pieces of down woody debris, or fine fuels (i.e., 1-, 10-, and 100-hour fuels), have the most influence over fire behavior because they have a large surface area compared to their low volume, thereby drying out and reaching ignition temperature more rapidly than larger fuels. Alternatively, coarse woody debris (CWD) (i.e. 1,000-hour fuels) are usually associated with fire effects (i.e., soil heating, emissions from combustion, etc.) because they generally burn longer in both the flaming and smoldering phases of combustion (Lutes and Keane 2006).

One of the most important consequences of bark beetle-caused tree mortality on fire behavior may be the reduction in sheltering that occurs with needle fall (Jenkins 2011). The opened canopy allows for greater solar insulation and drier fuels, and increased midflame wind speeds (Page and Jenkins 2007*b*). As fine fuels and windspeed increase, and foliar moisture content (FMCn) decreases during and after the epidemic phase, there can be an increase in fireline intensity, even under moderate fire weather conditions (Page and Jenkins 2007*b*). Fireline intensity is the rate of heat release from fire as determined by observations of flame length and can be used to predict the effect of fire on fuels in and above the flame (Rothermel and Deeming 1980). The probability of ignition beneath bark beetle-killed trees is thought to increase as the amount and depth of litter and fine woody fuel increases. However, Waldrop *et al.* (2010) experienced difficulty in igniting prescribed fires in SPB-killed areas within the Clemson Experimental Forest (CEF), Clemson, South Carolina if moderate moisture conditions existed or if canopy hardwoods shaded the burn unit.

Probability of ignition of CWD is greatly impacted by precipitation (amount and duration), and therefore fuel moisture content, but this relationship has not been well studied. One study conducted within the CEF aimed to determine how the moisture content for 100-hr and 1000-hr dead pines changed with rainfall. It was found that these large fuels were not impacted after a single rainfall event; however, several rainfall events over time produced increased fuel moisture (Mohr and Waldrop 2009). Probability of ignition was unknown, but if ignited, these fuels could amplify fire intensity and/or duration and become a problem for smoke management (Mohr and Waldrop 2009).

As snags begin to fall, heavy fuel loads accumulate on the forest floor. Waldrop *et al.* (2007) found that 1-, 10-, 100- and 1,000-hr fuels were significantly greater in SPB-killed stands than in non-affected stands within the Appalachian Mountains. Jenkins (2011) found that increasing fuel bed depths and CWD accumulation that occur after a beetle outbreak do not influence fire ignition or spread, but may add to surface fire intensity and energy release, especially during periods of drought. It was noted that the coarse woody fuel contained in standing snags may contribute to an increased period of flammability and fireline intensity when the site is shared with advanced regeneration in the decades following the outbreak. Historically, CWD fuels, standing snags, and advanced regeneration were consumed by high intensity wildfires, thus opening stands for the regeneration of pines. These high intensity wildfires had the potential to become crown fires when standing or leaning snags acted as ladder fuels to the forest canopy.

Although relatively few fires exhibit extreme fire behavior characteristics such as crowning (i.e., spreading from tree to tree within the overstory) (Pyne 1996), crown fires occur more frequently in the western U.S. than in the Piedmont and southeastern states. Active crown fires occur when effective canopy windspeeds are sufficient to move the fire from one tree crown to another. Active crown fires can occur in connection with an intense surface fire (dependent) or rarely without interacting with the surface fire (independent) (Jenkins 2011; Pyne 1996). Jenkins (2011) found that the following MPB-caused factors contributed to crown fire dynamics: crown base height, available canopy fuel load, foliar moisture content, and inter-crown distance. Simard *et al.* (2011) found that the transition from the red stage (current epidemic) to the gray stage (post-epidemic) in the early post-epidemic period may reduce the probability of active crown fire in the short term, primarily due to canopy thinning with needle fall. The role that needle terpenes may play on crown fire has not been well studied.

Plant terpenes are assumed to increase flammability of forest fuels, though few studies have documented this effect (Ormeño *et al.* 2009). It is unknown how much influence, if any, these compounds have on flammability; however, both increased terpene concentration and flammability were found in the litter of Mediterranean *Pinu*s species (Ormeño *et al*. 2009). Preliminary experiments in forest stands affected by MPB found that the level of terpenes decreased in whitebark pine (*P. albicaulis*) as needles changed from green to yellow, but increased in red needles (Jenkins 2011). Assuming they are flammable, an increase in terpenes in red needles would be expected to increase surface rate of spread and crown fire potential in these forests. No studies have researched the possible changes to terpenes that may take place in loblolly pine foliage during or after SPB activity.

Loblolly pine is considered fire resistant (Wade 1985) but not as resistant as longleaf pine or slash pine (Hare 1965). Mature loblolly can endure some fire defoliation and will survive low- to moderate-severity fires due to their relatively thick bark and tall crowns (Carey 1992). Loblolly pine's fire resistance increases with bark thickness and tree diameter, becoming resistant to low-severity fire by age 10 (Wahlenberg 1960). Needles are low in resin and not highly flammable (Landers 1991). Germination is epigeal and is enhanced by bare mineral soil (Carey 1992), with abundant regeneration occurring on soil exposed by fire (Brender *et al.* 1981). Where fire burns on average every 10 years, loblolly pine is considered a fire subclimax (Wahlenberg 1960), but in the absence of fire, loblolly pine is typically replaced by hardwood species.

Recently, there has been some scientific debate as to what extent beetle-killed forests affect wildfire. Intuition would tell us that the probability of extreme fire behavior will increase after an outbreak due to the high levels of fuel loading and decreased sheltering; however, recent evidence from western forest systems has been presented to indicate that there may be no difference or even dampened fire behavior in these areas. Romme *et al.* (1986*a*) found that while dead needles in the canopy may cause an increase in flammability 1-2 years after a MPB infestation, the risk of a destructive fire occurring during years 2-20 may decrease due to the reduction in continuity of canopy fuels and a small proportionate increase in forest floor fine fuels. Similarly, Simard *et al.* (2011) found that the transition from the red stage to the gray stage in the early post-epidemic period may reduce the probability of active crown fire in the short term, primarily due to canopy thinning with needle fall. Kulakowski *et al.* (2003) found that behavior of a wildfire in Colorado did not support the suggested increase of fire hazard in sub-alpine forest stands that had previously been affected by spruce beetle outbreak. They suggested that stands affected by beetle outbreak, regardless of the amount of downed woody fuel, may experience increased moisture due to the growth of understory herbs, and thereby decreasing the potential of low-severity fire to spread. Kulakowski and Jarvis (2011) found that climate and dry conditions were more important to Colorado and Wyoming lodgepole pine forest fire regimes than changes in fuels associated with MPB outbreak.

Evidence in support of the potential for extreme fire behavior in bark-beetle killed stands has also been documented in the recent literature. Page and Jenkins (2007*b*) indicate that surface rates of spread, fireline intensities, and crown fire potential were higher in current red stage MPB-killed lodgepole pine stands and gray stage stands than in endemic stands. They found that the high levels of tree mortality and dead canopy mixed with live foliage in epidemic stands could also increase crown fire potential. Klutsch *et al.* (2011) found increased intensity and smoke production in post-epidemic lodgepole pine stands. Schoennagel *et al.* (2012) found similar results, with increased intensity in grey stages and older MPB-killed stands than in endemic or current epidemic lodgepole pine stands. A study of potential fire behavior within spruce beetle-killed Intermountain spruce-fir stands found that post-epidemic, followed by epidemic and endemic stands, respectively, had faster rates of spread and higher fireline intensities (Jorgansen and Jenkins unpublished). Epidemic and post-epidemic classes also had longer smoldering duration and greater fuel consumption than endemic stands. Many studies with conflicting results of the effects of MPB outbreak on wildfire behavior have been conducted in the western U.S.; however, at the time of writing, this study will be the first to have been conducted on the effects of SPB outbreak on wildfire behavior in the eastern U.S. Given the great differences between vegetation, fire type, and fire behavior between eastern and western forest systems, the results of this study will provide valuable new information to resource managers within this region.

Decomposition and Resulting Succession of SPB-Killed Forest Stands

A series of ecological changes occur within affected forest stands during and after a beetle outbreak. After a small-scale canopy disturbance, dominant trees are removed, canopy openings are created, and there is canopy closure from adjacent trees, height recruitment of existing regeneration, establishment of regeneration, establishment or growth of shrubs and herbaceous plants, and generation of snags or large wood on the ground (Franklin *et al.* 2007),. In the case of forest stands attacked by SPB, this process begins with an individual tree infestation and subsequent death.

Within the weeks following initial attack, green needles begin to fade and turn yellow. The time required for needles to fade depends on several factors, including drought conditions and season. Pines attacked during the summer may fade in about two weeks, whereas trees attacked in the early spring or late fall may not fade for several weeks to months or may stay green even upon falling (Hyche 1999; Price 2008). By the end of the beetle’s life cycle (about 30 days), tree crowns are usually fully red (Hyche 1999). Depending on the degree of drought at the time that the tree is killed, red needles may stay on a loblolly pine for a few months to a year after death (John Nowak, personal communication). This red phase continues until all needles have fallen from the crown to the forest floor, at which time the tree is then considered to have entered the gray phase.

As individual trees are attacked during an outbreak, litter and fine woody fuels accumulate in pulses beneath their crowns. The rate of accumulation and spatial distribution of these fuels are dependent upon the arrangement and quantity of infested trees within the stand, as well as by crown conditions such as crown width, inter-crown distance, crown base height, and canopy bulk density (CBD) (Jenkins 2011). CBD decreases as the needles fall from the tree, later followed by branchwood. Litter ceases to accumulate when all needles have fallen off an individual attacked tree. Litter accumulation and decomposition achieve a balance as litter decomposes to duff within one to two years (Jenkins 2011). Fine woody fuels will continue to accumulate as small twigs are broken off from standing snags by snow or wind. Maximum duff depth and amount is reached the year following the end of needle fall (Jenkins 2011). Significant increases in needle litter amount and depth, and increases in 1-hr woody fuels occur between the MPB endemic and epidemic phases in lodgepole pine (Klutsch *et al.* 2009; Page and Jenkins 2007*a*). Jenkins (2011) found similar changes in whitebark pine fuels in Wyoming.

The accumulation of larger woody fuel varies considerably during the post-epidemic phase. Jenkins (2011) found that fuels larger than 1-hr did not increase significantly until well into the post-epidemic phase when large branches from standing snags fall to the forest floor. Over the course of several years, the remaining snag will break off in sections or fall over whole, sometimes piling one on another (Stottlemyer 2011). Nicholas and White (1984) found that wood debris weights were four times higher in previous outbreak areas than in unaffected stands, with total volumes of wood debris up to three times higher in SPB-infested versus non-infested stands. Despite the large volume of dead wood created by beetle-kill (Nicholas and White 1984; Stottlemyer 2011; van Hees 1992), very little is known regarding the natural fate of these dead trees.

The length of time that trees killed by bark beetles will remain standing varies considerably with topography (slope steepness, position, and aspect), soils, decay pathogens, exposure of the stand to wind, and the rate of decay of the tree's root system (Jenkins 2011; http://www.fs.fed.us/r3/resources/health/beetle/faq.shtml#content). A regression model created by Radke *et al.* (2009) indicated that approximately 57% of an average individual loblolly pine's stem would be classified as "down" within 2 years following death, and 90% would be down by 8 years (when using $a_2 = 0.724$ and $b_2 = 2.5$). A study by Moorman (1999) conducted from 1982-1990 within the Clemson Experimental Forest, SC found that 100% of loblolly snags in the study area were still standing one year after death; after 2 years 64% remained standing, 37% remained after 3 years, 20% remained after 4 years, 5% remained after 5 years, and 4% remained after 6 years. Other tree species contained similar rates of longevity. However, Holsten *et al.* (1995) found that as much as 50% of spruce beetle-killed spruce (*Picea* sp.) trees were still standing after 16 years within southcentral Alaska. Falling snags and the accumulation of CWD increases woody fuel bed depth (Jorgensen and Jenkins unpublished; Page and Jenkins 2007*a*). A balance between accumulation and decay is reached during the post-epidemic phase as CWD decomposes (Jenkins 2011). In this, the generality that downed woody fuels accumulate over time may not always be true (Brown and See 1981).

There is substantial variability in decomposition rates of pines killed by SPB. Decomposition begins after tree death and before the tree falls. After the tree falls, the chemical qualities of the habitat substrate affect the nutrients available for decomposition, as well as the culture medium for microorganisms. The process of decomposition is very complex, but it is generally accepted that decay is slower in larger diameter fuels due to a smaller surface to volume ratio. Decomposition of CWD is determined by a combination of species, size, chemistry, structure, soil fertility, the abiotic environment in which decomposition is occurring (temperature, aeration status, moisture), and a plethora of other site and environmental factors (Gambrell 2008; Waldrop 1996).

A model of long-term southeastern xeric and mesic mixed species stand dynamics after clearcutting predicted that differences in decomposition rates (possibly due to stand productivity) can be more important to fuel loading than fuel size (Waldrop 1996). Lutz spruce snags killed by the spruce beetle had very low rates of decomposition except at their bases; however, once the snags fell to the ground their decomposition rates increased and significant volume loss occurred (USDA Forest Service 2005*b*). Radke *et al.* (2009) found that residual loblolly CWD would be nearly completely decomposed after 25 years. Eventually the dead woody fuels deteriorate and settle over time, increasing compactness as supporting branches decay (Brown 1975). Page and Jenkins (2007*a*) found that small fuels within MPB post-epidemic stands decay sufficiently over time so that the differences return to original levels at least after 20 years.

Decomposition of dead trees and growing of successional species occur simultaneously post-outbreak. Upon falling, the dead trees create gaps in the forest canopy that allow increased light availability in the understory. Alterations in light availability may change overstory and understory species composition (USDA Forest Service 1989). The site will then become colonized by early successional vegetation including trees, shrubs, vines, and herbaceous plants, resulting in a thick, impenetrable tangle of piled logs and dense undergrowth (Stottlemyer 2011). During this stand initiation stage and subsequent stem exclusion stage, accumulation of CWD is slow since decomposition is occurring on the larger downed CWD and input rates are low due the small size of most of the dying vegetation (Waldrop 1996).

Successional vegetation in areas affected by disturbance is determined by the type of disturbance that occurred. In the case of insect outbreaks, including those of SPB, the damage is primarily to large trees, which may accelerate the process of succession (Frelich 2002; Lafon and Kutac 2003, Waldron *et al.* 2008). A comparative study of SPB and non-SPB impacted forests of the USDA Forest Service Coweeta Hydrologic Laboratory near Franklin, NC found that there was greater tree seedling species richness in SPB-damaged plots than in unaffected plots, while herb layer species richness tended to be greater in unaffected plots (Kloeppel *et al.* 2004). Additional studies conducted at the Coweeta Hydrologic Laboratory found that on sites where no human intervention occurred, SPB-affected stands that were previously pine-dominated became occupied by oaks and low-quality hardwoods (Smith 1991) with a dense mountain laurel understory (Vose and Swank 1993). The forest stands in the present study were dominated by loblolly pine prior to outbreak. Loblolly pine is moderately tolerant of shade when young but becomes intolerant with age. Its rapid growth allows it to dominate a site early (Baker and Langdon 1990), often invading old fields, clearcuts, and other disturbed sites. However, in the absence of major disturbance, loblolly pine stands are gradually replaced with hardwoods or a mixture of hardwoods and pines.

Stand density and basal area of host trees decrease following a bark beetle outbreak. However, Romme *et al.* (1986*b*) found that surviving trees in the canopy, sub-canopy, and understory grew more rapidly after an MPB outbreak than they did prior to outbreak. These surviving trees also had higher growth ratios than trees in nearby control stands. Gaps created by falling trees provide increased availability of light, water, and nutrients, allowing for greater productivity in the remaining trees. Waring and Pitman (1985) found that tree growth in post MPB-outbreak stands increased primarily due to improved light availability. A silvicultural treatment study in the coastal plains of Georgia and North Carolina found that loblolly pine seedlings experienced increased growth when located in gaps created by silvicultural treatments such as patch cutting or clearcuts (Knapp *et al.* 2011). Similar increases in growth of surviving trees following the death, defoliation, or removal of a portion of the canopy have been described in other coniferous forests from many parts of the United States (Ferguson and Adams 1980; Ferrell 1980; Gordon 1973; MacLean 1984; McCaughey and Schmidt 1982; Moore and Hatch 1981; Seidel 1980, 1983). Very few studies have attempted to quantify the vegetative regeneration that takes place after SPB outbreak. The results of this study will provide a better understanding as to the type and amount of regeneration that occur within SPB post-outbreak forest stands over time.

Modeling Fire Behavior

Fire is dynamic, with the size, rate, and spread being dependent on a host of variables, including fuel loading, weather, terrain, etc. Wildland fuelbeds have spatially and temporally complex structures, geographic and physiographic differences, and contain components with varying degrees of flammability. Since it would be difficult to inventory each of these characteristics every time an assessment or management decision was necessary (Sandberg *et al.* 2001), modeling is often used instead. A large number of variables must be considered before modeling can take place (i.e., types of deadwood, limb structure of trees, moisture content of living leaves, moisture content of dead grasses and twigs, weather, terrain, etc.) (Andrews 2007). In addition, the type of fire you are trying to model (i.e., crown vs. surface) will play a role in the data necessary for input.

A mathematical fire model refers to a set of algorithms that describe a single physical trait of a fire, such as surface fire spread, flame length, or reaction intensity. Computer programmers then combine multiple models to create a fire-modeling system that can be run on a computer to produce fire forecasts and maps. Fire modeling systems can be used by land managers, fire fighters, fire danger analysts, or researchers to successfully predict wildland fire behavior and fire effects. These predictions can be applied to a range of fire management activities including wildfire behavior, prescribed fire planning, fuel hazard assessment (Andrews 2007), and carbon accounting (Johnsen *et al.* 2001).

Because fire has become so important to land management and there is an ever-increasing urban/wildland interface, numerous fire modeling programs have been created to date. One such program, the Fuel Characteristic Classification System (FCCS), is a software application used to compare potential fire behavior between varying fuelbed characteristics and changing weather conditions (Prichard *et al.* 2011). Users may select from over 200 FCCS fuelbeds to represent their study area, or can modify and enhance an existing fuelbed for customization to reflect actual site conditions. The existing fuelbeds represent common fuel types in North America and were created based on scientific literature, fuel photo series, fuels data sets, and expert opinion (Ottmar *et al.* 2007). FCCS was designed to give users the ability to represent the structural complexity and diversity of fuels produced by natural processes (including insect outbreak) and various management activities. It allows users the opportunity to create and catalog fuelbeds and to classify those fuelbeds based on their capacity to support surface and crown fires and to consume and smolder fuels (Ottmar *et al.* 2007; Sandberg *et al.* 2007*a*).

FCCS fuelbeds are organized based on seven qualitative criteria, primarily by Bailey's ecoregion division (Bailey 1989) and vegetation form (i.e., conifer forest, hardwood forest, grassland, etc.). The fuelbeds are then categorized into six horizontal strata: canopy, shrubs, non-woody vegetation, woody fuels, litter-lichen-moss, and ground fuels. These strata were chosen by FCCS developers as an attempt to represent every fuel element that has the potential to combust, and are further broken down into subcategories in some instances (Prichard *et al.* 2011). Environmental variables including fuel moisture, midflame windspeed, and slope gradient are also included to account for changes in weather and topography. Users can choose to use existing fuelbeds and environmental variables or to customize them using their own data. Once all qualitative edits have been made, FCCS calculates and generates results based on the input information. Several reports, including quantitative fuel characteristics (physical, chemical, and structural properties) and potential surface fire behavior specific to the fuelbed are calculated (Riccardi *et al.* 2007*a*).

Surface fuelbed data are useful for predicting fire spread and intensity, either directly, by using a reformulated Rothermel (1972) fire spread model created by

Sandberg *et al.* (2007*b*), or indirectly by providing a “crosswalk” from a fuelbed to one of 13 original fire behavior fuel models described by Anderson (1982) and one of 40 standard fuel models described by Scott and Burgan (2005). To facilitate use in modeling systems, fuelbed inputs have been formulated into fuel models. A fuel model is different from a mathematical fire model or a fire modeling system; a fuel model is a set of fuelbed inputs needed by a particular fire behavior or fire effects modeling system and is a set of values that describe the surface fuel as required by the Rothermel (1972) surface fire spread model. The 13 original fire behavior fuel models describe common fire-prone fuel types (i.e. grasslands, brush, timber, slash, etc.); the additional 40 fuel models (Scott and Burgan 2005) expand on the original models and provide more fuel type options such as forest litter, litter with grass or shrub understory, or fuels in high-humidity areas.

According to an FCCS developer, the crosswalk from FCCS to a fuel model is only based on predicted surface fire flame length and rate of spread, and does not correlate with vegetation or other input variables (Susan Prichard, personal communication). The crosswalk is static, being valid only under the set of environmental variables (fuel moisture, midflame windspeed, and slope gradient) that were specified (Prichard *et al*. 2011). The primary purpose of the crosswalk is to provide the closest fuel model match in terms of predicted surface fire behavior (Susan Prichard, personal communication). The fuel models resulting from this crosswalk can then be used within fire modeling systems that require their use for fire behavior prediction, such as BehavePlus.

The BehavePlus fire modeling system is one of the most commonly used fire behavior prediction systems for wildland fuels. BehavePlus is a flexible fire modeling system developed by USDA Forest Service researchers that produces tables and graphs used for a multitude of fire management applications including projecting the behavior of an ongoing fire, planning prescribed fire, and training (Andrews *et al.* 2008). The original BEHAVE fire behavior prediction and fuel modeling system was created to predict fire spread and intensity (Andrews 1986; Andrews and Bradshaw 1990; Andrews and Chase 1989; Burgan and Rothermel 1984), and was among the first computer systems developed for wildland fire management (Andrews 2007). Five later versions (BehavePlus v.1-5) were developed as successors to BEHAVE to incorporate updates in user needs and in fire and modeling features and capabilities.

The fire modeling capabilities of BehavePlus v.5 are grouped into the following nine modules: surface, crown, safety, size, contain, spot, scorch, mortality, and ignite. Each module is comprised of several models (mathematical relationships that describe a specific aspect of the fire). There are approximately 35 models used in BehavePlus (Andrews 2007), some of which are categorized as fuel models. Of the 13 original (Andrews 1982) and 40 additional (Scott and Burgan 2005) fuel models used within BehavePlus, the most commonly used fuel model for the Piedmont is ‘timber (pine and hardwood) litter’ (TL9). This fuel model is described as having a very high load of dead and downed woody fuel (litter) beneath a forest canopy with moderate spread rate and flame lengths (Scott and Burgan 2005). Studies using BehavePlus for coniferous forests in the Piedmont are rare, but have been presented by Mohr and Waldrop (2006) and

Stottlemyer (2011). The surface module uses the following variables for calculation to predict surface fire behavior: surface fire rate of spread, fireline intensity and flame length, reaction intensity and heat per unit area, intermediate values (i.e., characteristic dead fuel moisture, relative packing ratio, etc.), and environmental variables (i.e., wind speed and direction, slope gradient, and fuel moisture) (Andrews 2008).

In order to reflect site-specific environmental conditions by customizing wind speed, slope gradient, or fuel moisture within FCCS or BehavePlus, users either need to collect this data from their study site, or obtain available environmental data from the region. Although several default moisture scenarios containing moisture conditions of the surface fuel for 1-, 10-, and 100-hour, as well as herbaceous and woody fuels are provided within both FCCS and BehavePlus, they are not specific to a particular site or region. Using customized moisture conditions rather than default values provide more accurate and useful fire behavior predictions. FireFamily Plus (FFP) can be used to generate weather and fuel moisture conditions from a specific site or region when creating custom moisture scenarios. FFP is a software system used to summarize and analyze historical daily fire weather observations and compute fire danger indices based on the United States National Fire Danger Rating System (NFDRS) (Bradshaw and McCormick 2009). Historical weather data from weather stations can be obtained to determine percentile weather and fuel moisture data for a specified period of time. Results can be used to compare historic weather during fire season and non-fire season, set threshold levels for fire management actions, such as forest restrictions, viable areas for non-suppression, and more. This information is useful in fire prediction modeling systems when comparing fire behavior at various environmental conditions.

Not all researchers feel that modeling programs accurately predict fire behavior within beetle-killed stands. Jenkins (2011) indicated that fire prediction systems did not accurately handle inputs of live shrubs and forbs below a MPB-killed conifer stand during epidemic and post-epidemic phases. Since many shrub and forb layers have high live fuel moisture contents, he believed they would act as a "wet blanket" over the litter and DWD that had accumulated after outbreak. In this, a "typical" conifer litter understory fuel model may predict a higher probability of ignition and surface fire spread rate than what will actually occur. It was noted that the considerable variability in herbaceous plant and shrub composition and flammability makes the beetle-killed fuel complex even more difficult to model.

Even though fuel models exist for predicting fire spread in slash-blowdown fuel types, Stottlemyer (2011) felt that input values for fuel loading may not accurately characterize the beetle-killed fuel complex. When comparing fuel loads from SPB-killed stands to the fuel loads outlined in the fuel models described by Scott and Burgan (2005), Stottlemyer (2011) found that the slash-blowdown model 'Low Load Activity Fuel (SB1)' had input values that were most appropriate, but overestimated fuelbed depth as well as biomass for 1- and 100-hr fuel loading. Until standard fuel models which can more accurately predict fuel loading and fire behavior for these post-outbreak stands are created, customized fuel models will need to be tailored to fit individual stand conditions for those sites located within SPB-killed areas.

CHAPTER THREE

METHODS

Field measurements and modeling were combined to study fuel dynamics and potential fire behavior within control and post-outbreak loblolly pine-dominated forest stands across a chronosequence ranging from 0 to 8 years since outbreak. To obtain baseline measurements of live and dead fuels in these SPB-killed areas, fuels were measured within the Clemson Experimental Forest, Oconee National Forest, and Sumter National Forest, USA using planar intersect methodology (Brown 1974). Fuels in control stands were compared to stands killed by SPB at 2 and 8 years post-outbreak so that fuel dynamics could be characterized in a time-since-SPB-outbreak chronosequence. The Fuel Characteristic Classification System (FCCS) was used to construct a representative fuelbed for each stand using the inventoried data. Fire behavior was then modeled using both FCCS and BehavePlus to understand the consequences of the changes in fuel between control and post-outbreak stands. FireFamily Plus was used to generate 80^{th}, 90^{th}, and 99^{th} percentile fuel moisture conditions, and results were subsequently used within FCCS and BehavePlus to observe the effect of fuel moisture conditions on predicted fire behavior.

Study Sites

Fuels data were collected at 42 forest stands within three study sites located in the Piedmont of Georgia and South Carolina (Figure 1). These study sites were assumed to

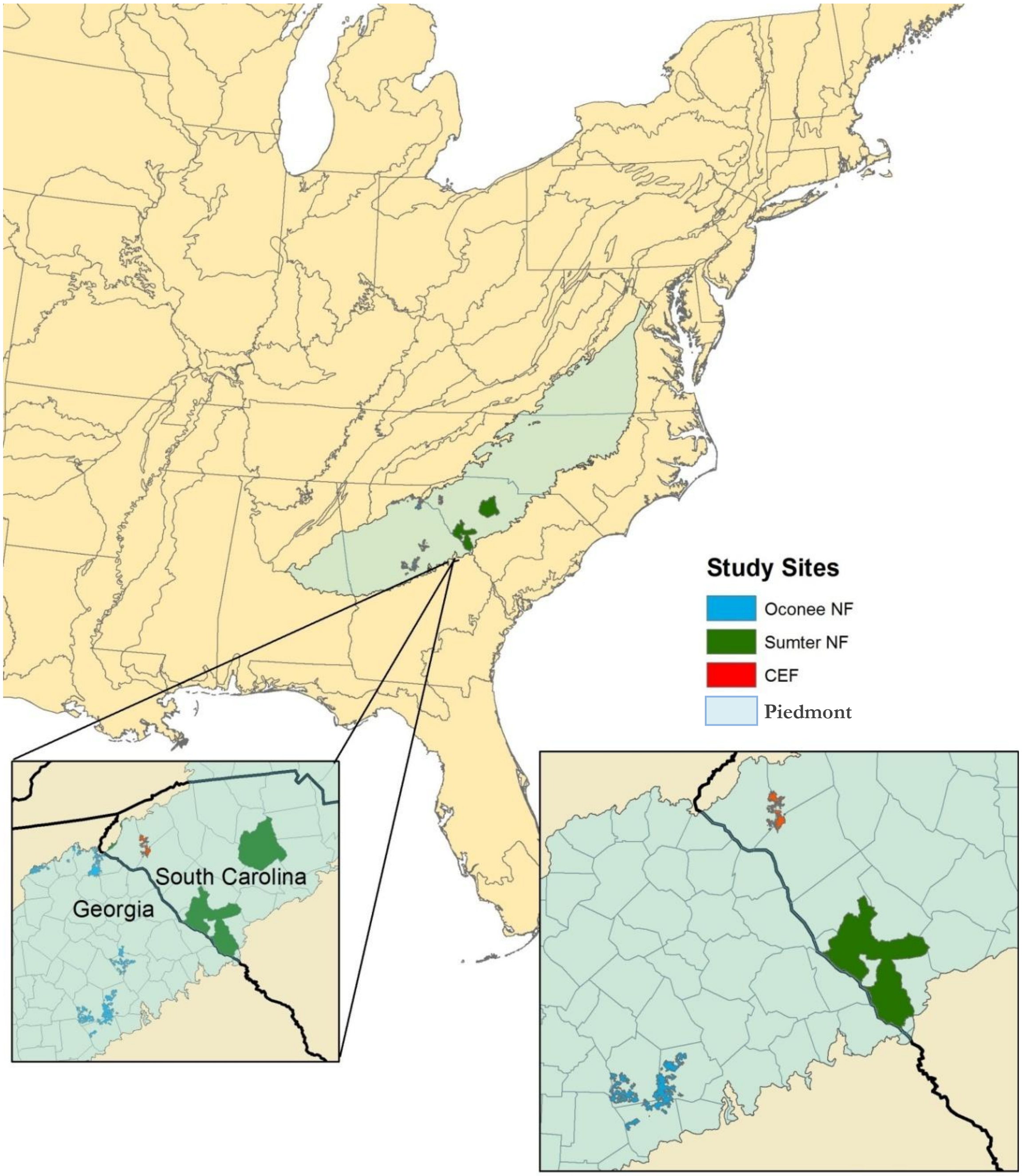

Figure 1. Location of study sites where fuels data were collected: Clemson Experimental Forest (CEF), Long Cane Ranger District of Sumter National Forest (SNF), and Oconee Ranger District of Oconee National Forest (ONF).

be largely representative of similar forest stand types within the Piedmont physiographic province. For purposes of this study, the Piedmont was defined using the ecoregion framework originally developed by Omernik (1987) and later updated by the U.S. Environmental Protection Agency (EPA) (1999). Physiographically, the Piedmont is considered a province of the larger Appalachian Highlands physiographic division. Its eastern boundary is separated from the Coastal Plain by the fall line and is mostly bounded by the Blue Ridge Mountains to the west. The surface relief of the Piedmont is characterized by relatively low, rolling hills with heights above sea level between 200 to 1,000 feet. The land within the Piedmont was once extensively farmed, resulting in soils that are acidic, clayey, and low in fertility. Since a majority of the top soil has eroded, production rates are low. Land cover patterns today consist of a mixture of forest, agriculture, and urban development. The Piedmont contains portions of Virginia, North Carolina, South Carolina, Georgia, and Alabama.

The first study site was located within the 115,353-acre Oconee National Forest (ONF) (33°24’48” N, 83°22’21” W), which is about halfway between Macon and Athens, Georgia. The ONF is spread over eight northern Georgia counties and is organized into one ranger district (Oconee Ranger District). The ONF consists of nearly flat terrain with small hills ranging from 277 to 894 feet above sea level. The uplands commonly contain well-drained, very gently sloping to strongly sloping soil series including Cecil, Pacolet, Hiwassee, and Wedowee (http://www.fs.usda.gov/detail/conf/landmanagement/planning/?cid=fsm9_029149). Mean annual temperature is 63.8°F, with mean temperatures ranging from 45.3°F in

January to 80.6°F in July, and mean annual precipitation of 45.84 inches distributed fairly evenly throughout the year, with April, May, and October being the driest months (USDC NOAA 2002*b*). Several SPB outbreaks have taken place here in recent years, with the most recent large-scale outbreak occurring from 2007–2008.

The second study site was located within the 120,000-acre Long Cane Ranger District (LCRD) of the Sumter National Forest (SNF), SC (34°01′17″ N, 82°15′59″ W). The LCRD is characterized by rolling hills of moderate relief and elevations ranging from 394 to 591 feet above sea level. The area is dominated by the Cecil soil series (fine, kaolinitic, thermic Typic Kanhapludults), characterized as "very deep, well-drained moderately permeable soils on ridges and side slopes formed in residuum weathered from felsic, igneous and high-grade metamorphic rocks" (USDA Soil Survey Staff 2007). Climate at the LCRD is mild, with mean temperatures ranging from 41°F in January to 78.8°F in July, and mean annual precipitation of 46.32 inches distributed fairly evenly throughout the year (USDC NOAA 2002*a*). In recent years, the most destructive SPB outbreaks took place from 2001–2002 and 2007–2008.

The third study site was located in the Clemson Experimental Forest (CEF), within Pickens County in northwestern South Carolina (34°40'N, 82°49'W). Topography of the 17,356-acre CEF is characterized by slightly to moderately steep hills and slopes, with elevations ranging from 698 to 1,099 feet above sea level. The dominant forest type is loblolly pine and shortleaf pine (*Pinus echinata* Mill.), most of which is second- or third-growth timber resulting from reforestation programs of abandoned agricultural fields in the early 1900's. The parent material of the soil consists of phyllites, granites, gneisses, and various schists formed in the late Precambrian to early Paleozoic age (Sorrels 1984). The clayey, well drained soils are generally classified as Typic Kanhapludults or Typic Hapludults, with the majority falling into Pacolet, Madison, Cecil, and Tallapoosa soil series (Shelburne *et al.* 2004). Climate at the CEF is mild, with mean temperatures ranging from 41°F in January to 78.8°F in July (USDC NOAA 2002*a*). Mean annual precipitation is 54.0 inches, with January to March containing the most rainfall and the rest distributed fairly evenly throughout the year (USDC NOAA 2002*a*). Several SPB outbreaks have occurred here in recent years, the most destructive of which took place from 2001–2002.

Plot Establishment and Fuel Sampling

Forest stands killed by SPB outbreak at different years were identified based on existing records and aerial photos. Twenty-six affected forest stands were then chosen *a priori* based on their location within the Piedmont, dominance of live or dead loblolly pine, and presence of previous SPB outbreak. In addition, 16 control stands (i.e., not affected by SPB) were identified (Table 1). The stands formed a chronosequence consisting of three distinct age classes from time of outbreak, ranging from 0 years (control stands) to 2 years (killed in 2007) (i.e., early post-outbreak) to 8 years since outbreak (killed in 2002) (i.e., late post-outbreak). Locations of control stands were chosen based on proximity to SPB-killed stands, dominance of live loblolly pine, and containing no observable evidence of previous SPB outbreak. When possible, control plots were located within the same stand as the post-outbreak plots so that differences in

Table 1. Number of stands, plots, and transects measured at each of the three study sites: Clemson Experimental Forest (CEF), Sumter National Forest (SNF), and Oconee National Forest (ONF). Early post-outbreak (EPO) stands were killed in 2007, while late post-outbreak (LPO) stands were killed in 2002.

Location	No. of Control Stands	No. of EPO stands	No. of LPO stands	Total No. of Stands	Total No. of Plots	Total No. of Transects
CEF	3	0	3	6	24	72
SNF	6	7	9	22	88	264
ONF	7	7	0	14	56	168
TOTAL	16	14	12	42	168	504

fuel loading and stand structure could be attributed to beetle kill, rather than variations in precipitation, wind, or season. The study design at the ONF and CEF was a randomized complete block. The ONF had 7 blocks, each with one control plot and one early post-outbreak plot. The CEF had 3 blocks, each with one control plot and one late post-outbreak plot. The SNF used an unbalanced, completely random design with measurements occurring in 16 separate stands. Three stands contained a control plot and an early post-outbreak plot; 3 stands contained a control plot and a late post-outbreak plot; 4 stands each had an early post-outbreak plot and no control plots; and 6 stands had a late post-outbreak plot and no controls. In this paper, the stand types are often referred to by their initials as follows: early post-outbreak (EPO), and late post-outbreak (LPO).

Fuels data were collected from September 2009 to January 2010. On each identified stand, down woody debris (DWD) was measured using the Brown's planar intersect method (Brown 1974). Three 50 foot transects were established at each of four randomly-selected plots within each stand. Plots located within the same forest stand were located at least 150 feet apart. Orientation of the center transect was established by placing the zero end of a measuring tape in a randomly-selected direction. Two additional measuring tapes were then anchored at the same location and placed outside of the center transect, forming a 45 degree angle from one another (+22 degrees and -23 degrees from the center transect, respectively) (Figure 2). Measurements along the fuel transect began at the common point for the center transect, and on the end away from the common point for the outer transects. This design has been used in several other studies, including Waldrop *et al.* (2010).

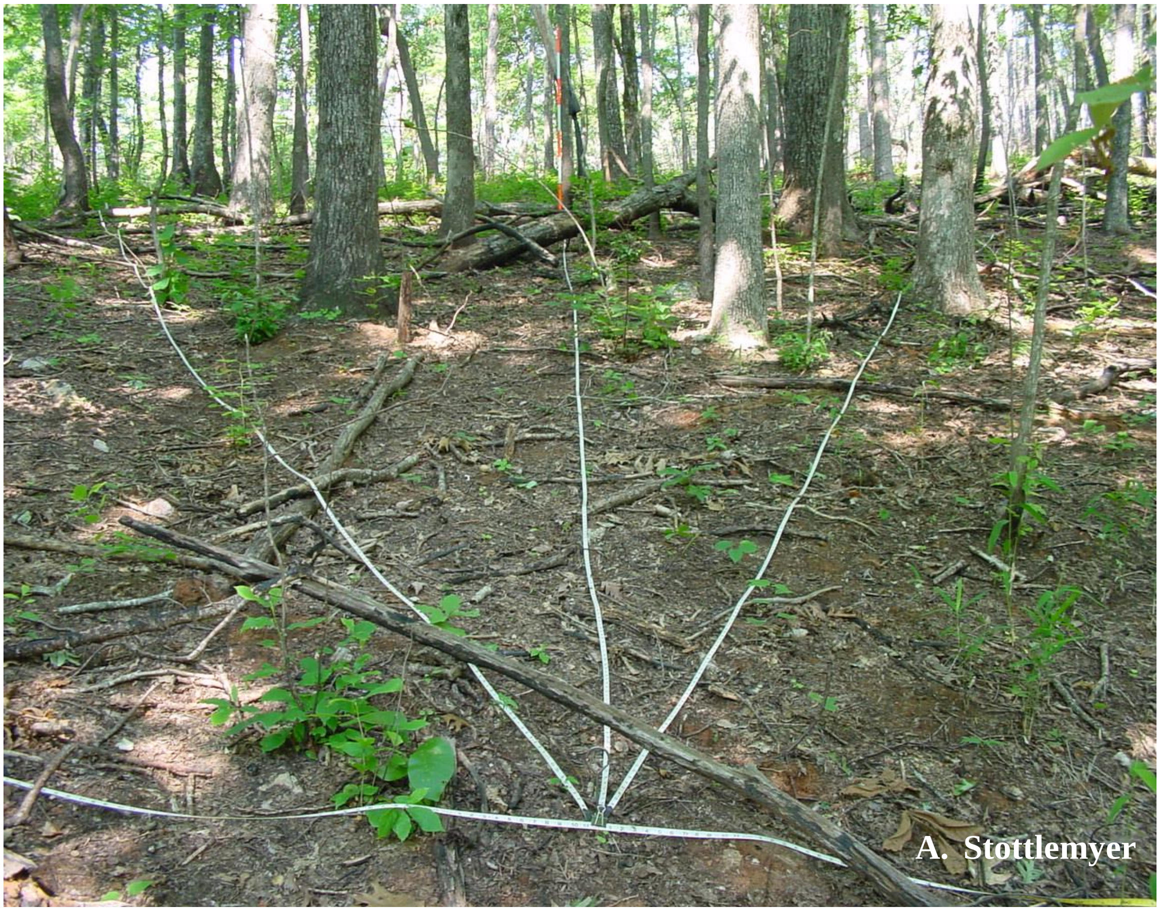

Figure 2. Planar intersect methodology used to collect downed woody fuels data within three Piedmont study sites, 2009 – 2010.

Fuels intersecting the sampling plane were tallied in the standard fire size classes: 1-hour (0–0.25 inches in diameter), 10-hour (0.25–1 in), 100-hour (1–3 in), and 1,000-hour fuels (>3 in). One- and 10-hour fuel intercepts were counted along the first 6 feet and 100-hour fuel intercepts were counted along the first 12 feet. Fuels in the 1,000-hour class were recorded by species and diameter, and were categorized into one of five decay classes following the classification outlined by Pyle and Brown (1998) along the entire 50 foot transect. If these fuels could not be identified to species they were recorded as pine or hardwood. Trees no longer containing measurable CWD due to advanced stages of decomposition were not included.

Aboveground height of dead and down wood were measured along 1 foot sections beginning at 12, 25, and 40 feet. Fuel height was measured from the surface of the mineral soil to the highest dead and down woody fuel particle (<6 ft) which intersected the transect (Brown 1974). Litter and duff depths were measured to the nearest 1 mm, and percent grasses, forbs, and woody vegetation cover were visually estimated at 12, 25, and 40 feet along every transect. Basal area of standing live and dead trees was measured using a 10-factor prism, and recorded by species. To obtain sapling density, live trees were measured by species and diameter at breast height (DBH)(4.6 ft above ground) along 6.6 feet from each side of the center transect, thus forming a 13.2 x 50 foot (0.015 ac) subplot. Counts within subplots were tallied into three size classes: 0–0.5 inches, 0.5–1.5 inches, or greater than 1.5 inches in diameter. In addition, aspect, percent slope, and transect azimuth were also collected for each transect. Counts of 1-, 10-, 100- and 1000-hour fuels obtained from transect sampling in the field were converted to weights using equations given by Brown (1974) and specific gravity estimates for southern species by decay class developed by Anderson (1978).

Fuels data were collected from control and post-outbreak stands for each study site. Seven control and seven EPO stands were surveyed within the ONF. Within the SNF, six control and 16 post-outbreak (EPO and LPO) stands were surveyed. Within the CEF, three control and three LPO stands were surveyed. All of the affected stands were approximately one hectare in size, while the size of control stands varied. The stands were plantations and were approximately 18–31 years in age when killed (Knight Cox, personal communication).

Additional Data Used for Analyses

All fuels data used for analyses and modeling were obtained from fuels measured at the study sites. Any additional information necessary for modeling purposes were generated based on data obtained from other sources. For example, both of the fire behavior modeling programs used in this study required fuel moisture inputs in order to create custom moisture scenarios; however, since no fuel moisture data were collected in the field as a part of this study, this information was derived from FireFamily Plus (a software system that summarizes historical daily fire weather observations and generates fuel moistures). Fuel moisture plays a large role in predicted fire behavior; therefore, accurate fuel moisture inputs typical of this region were necessary for modeling purposes.

In order to simulate burning conditions most typical of a Piedmont wildfire, only weather data from the fire season (i.e., not year-round) were used for analysis. For purposes of this study, the Piedmont wildfire season was considered to be when most wildfires (human or naturally-caused) occurred in the Piedmont in any given year. Although the South Carolina Piedmont wildfire season has generally been considered to be February through early April (Shelburne *et al.* 2004), few to no studies have provided empirical data which summarizes historic fire occurrences throughout the Piedmont ecoregion. Therefore, a large-scale, high-resolution federal wildfire occurrence dataset containing the dates and locations of historic wildfires within the United States between 1970 and 2006 was used to determine the Piedmont fire season. This dataset was originally compiled by the Desert Research Institute (DRI) to provide an inventory and quality control of occurrence records for the DRI Climate, Ecosystem, and Fire Applications (CEFA) program. Brown *et al.* (2002) provides details as to its compilation and an assessment of the quality of the data.

The wildfire occurrence dataset detailed the date, location, size, and cause of historic federal wildland fire occurrences. Only records deemed “usable” according to Brown (2002) (i.e., containing correct information according to quality control procedures) were used for this analysis, resulting in n=664,942 federal wildland fire occurrence records within the United States. In order to find only those occurrences located within the Piedmont, fires were segregated according to their spatial location using Arc GIS (v.10.0 Redlands, CA) by using latitude and longitude parameters provided within the dataset metadata. These points were then “clipped” to the Piedmont ecoregion boundary, reducing the dataset to n=3,714 occurrences. Within the Piedmont, the most fires were recorded in Alabama (n=1,328), followed by Georgia (n=1,083), South Carolina (n=759), North Carolina (n=402), and Virginia (n=142), respectively. The dates of these fires were then analyzed to determine the Piedmont fire season.

Based on the historic fire occurrence dataset, it was found that there were two fire seasons within the Piedmont, with a majority of fires occurring between the dry late winter/early spring months of February–April, and with a second smaller peak in November (Figure 3). The largest percentage of fires was documented in March for Alabama (18%), Georgia (19%), North Carolina (26%), and Virginia (18%), and in April for South Carolina (19%). The lowest number of fires occurred from May to September. Based on these results, the fire season within the Piedmont to be used for analysis was determined to be November 1 through April 30.

Fire Behavior Modeling

Data analyses and fire behavior modeling required the use of three computer programs. FireFamily Plus (FFP) (v. 4.0.2 Missoula, MT) was used to analyze historical daily fire weather observations within the Piedmont to obtain percentile fuel moisture conditions for use in creating custom fuel moisture scenarios within the fire behavior modeling programs. FCCS and BehavePlus were used to model the differences in fire behavior between control and post-outbreak stands. Details regarding each of these programs are provided below. Alternative fire behavior prediction modeling systems were considered when deciding which modeling programs to use for purposes of this study; however, other models do not accurately predict results for the Piedmont and/or were not applicable to this study. FOFEM, the First Order Fire Effects Model, contains

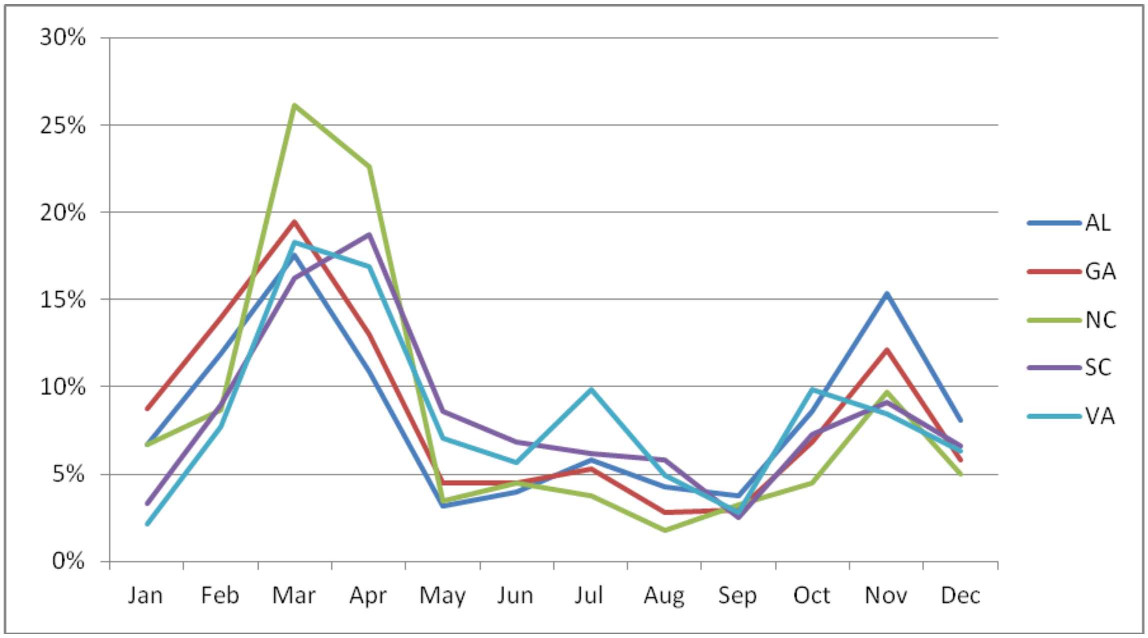

Figure 3. Percentage of fires occurring each year by month based on historic wildland fire occurrence data (n=3,714) within the Piedmont ecoregion from 1970 – 2006. This graph was created based on information derived from a federal wildfire occurrence dataset originally compiled by the Desert Research Institute. Fires were predominantly human-caused.

models based on western conifer forests and appears to be a poor predictor of mortality of southern pines in the southeast (Waldrop *et al.* 2006). Similarly, the use of the Fire Area Simulator (FARSITE) has grown in the southwest, midwest, Florida, and in other countries, but has been used infrequently in the Eastern United States (Waldrop *et al.* 2006). FARSITE was not accurate in predicting fire behavior for hardwood forests in the Southern Appalachians for most scenarios (Phillips *et al.* 2006). Several other models were not suitable for use in this study due to inappropriate fuel type or irrelevant outputs that were not useful to this study.

FireFamily Plus

To customize fuel moisture conditions within FCCS and BehavePlus, environmental data from the region were obtained using FireFamily Plus (FFP) (v. 4.0.2 Missoula, MT). FFP is a software system used to summarize and analyze historical daily fire weather observations. Further description of this program can be found in the 'modeling fire behavior' section of Chapter Two. FFP has been used in a multitude of fire-related studies and is commonly used to obtain historic fire weather information for modeling purposes. In this study, FFP was used to generate weather and fuel moisture conditions from weather stations located within the Piedmont. Weather and percent fuel moisture data inputs were necessary for fuel moisture scenario customization within FCCS and BehavePlus to reflect site conditions as accurately as possible. To obtain this information, historic weather station data (i.e., temperature, relative humidity, wind speed, precipitation) were first downloaded from the National Wildfire Coordinating

Group (NWCG) Fire and Aviation Management Web Applications (FAMWEB). The Shoal Creek (Alabama) (ID # 012902), Oconee #1 (Georgia) (ID# 093701), Whitmire (South Carolina) (ID# 380902), and Uwharrie (Troy) (North Carolina) (ID# 317001) Remote Access Weather Station (RAWS) were used for analysis. These weather stations were chosen based on proximity to the study sites, and were all located within the Piedmont ecoregion. Since Virginia had no RAWS located within the boundaries of the Piedmont, none were used in Virginia for analysis.

Since fuel moisture conditions play a large role in determining surface fire severity and intensity, it was important to use various moisture conditions during modeling to illustrate the effect of changing moisture conditions on fire behavior. Historic weather data were analyzed using FireFamily Plus in order to determine 80th, 90th, and 99th percentile weather and fuel moisture (1-, 10-, 100-, and 1000-hour) thresholds to use in FCCS and BehavePlus simulations. The 80th, 90th, and 99th percentiles were chosen to represent dry, very dry, and extremely dry conditions, respectively. These particular percentiles were chosen based on similarity to use within other fire behavior studies (Black and Opperman 2005; Schmidt *et al*. 2008; Schoennagel *et al*. 2012). The percentile weather and moisture conditions were obtained for the entire Piedmont fire season (November 1–April 30); analyses were run for the years 1965–2011. Results were subsequently used to create custom moisture scenarios within FCCS and BehavePlus.

Fuel Characteristic Classification System

FCCS (v. 2.2.1 Seattle, Washington) was used to model potential surface fire behavior for each of the study sites. The program was obtained from the U.S. Forest Service, Pacific Northwest Research Station, Fire and Environmental Research Applications Team (FERA) research/studies website. FCCS was chosen for this study based on its capability to model surface fire behavior using measured site-specific fuelbed properties and environmental conditions. It provides an objective comparison of fuelbeds by eliminating the subjectivness associated with having to choose or adjust existing fuel models as in other modeling systems.

The FCCS defines a fuelbed as "a relatively homogenous unit on the landscape, representing a unique combustion environment that determines potential fire behavior and effects" (Ottmar *et al.* 2007). FCCS v.2.2.1 provides over 200 fuelbed options with default parameters, however, the fuelbeds used in this study were customized to reflect actual site conditions. Plots within each control stand were relatively homogenous, dominated by loblolly pine in the canopy and containing similar mean fuelbed attributes (litter and duff depths, fuel loads, etc.). Similarly, the plots located within their respective SPB-killed stands were also relatively homogenous within each stand, containing similar vegetation and like mean fuel loads. However, in order to capture the possible heterogeneity between stands within and between stand types (i.e., control, EPO, and LPO), custom fuelbeds were created on a stand basis within each study site, for a total of 42 custom fuelbeds (n=16 control, n=14 EPO, and n=12 LPO).

Within FCCS, fuelbeds are organized based on seven qualitative criteria. Two of these criteria (i.e., ecoregion and vegetation form) are required for initial custom fuelbed creation, while the other five criteria (i.e., structural class, cover type, change agent, natural fire regime, and Fire Regime Condition Class) are optional, but can assist users in choosing fuelbeds more specific to their particular study area. All custom fuelbeds in this study were created using the Bailey's Subtropical 230 ecoregion in order to encompass the locations of the study sites and the Piedmont ecoregion. Based on overstory dominance at the time of data collection, the 'Conifer Forest' vegetation form was chosen for the control and EPO stands, while the 'Mixed Forest' vegetation form was used for the LPO stands. From the list of available fuelbeds that were then produced by FCCS based on these ecoregion and vegetation form inputs, 'loblolly pine forest (no change agent)' was chosen as the default fuelbed for the control stands and EPO stands. Not having any change agent associated with the fuelbed implies that no natural or management activities have altered the fuelbed. 'Loblolly pine-shortleaf pine- mixed hardwood forest (insects and disease change agent)' was selected as the default fuelbed for the LPO stands. The 'insects and disease' change agent implies that mortality of trees and other vegetation from insect attacks or disease has led to accumulation of dead fuels.

Parameters were customized for modeling within FCCS (Table 2). Input values were obtained from field data when possible. Based on recommendation by an FCCS developer, mean values, rather than mode (as the FCCS User Guide suggests), were used

Table 2. Parameters used in custom fuel models within FCCS for three Piedmont study sites.

Parameter	Units	Value
Canopy fuels		
Trees: live trees		
Overstory density	#/ac	field
Overstory species and relative cover	%	field
Understory density	#/ac	field
Understory species and relative cover	%	field
Snags: standing dead trees >1.4m		
Density	#/ac	field
Species and relative cover	%	field
Surface fuels		
Shrubs	% cover	field
Non-woody fuels	% cover	field
All woody fuels		
Fuel bed depth	ft	field
Sound woody fuels		
1-hr time lag fuel load	tons/ac	field/regression equation
10-hr time lag fuel load	tons/ac	field/regression equation
100-hr time lag fuel load	tons/ac	field/regression equation
1000-hr time lag fuel load	tons/ac	field/regression equation
Species and relative cover	%	field
Rotten woody fuels		
1000-hr time lag fuel load	tons/ac	field/regression equation
Species and relative cover	%	field
Litter depth	in	field
Duff depth	in	field
Fuel moisture		
1-hr time lag fuel moisture	%	percentile data
10-hr time lag fuel moisture	%	percentile data
100-hr time lag fuel moisture	%	percentile data
1000-hr time lag fuel moisture	%	percentile data
Non-woody moisture	%	percentile data
Shrub moisture	%	percentile data
Crown moisture	%	120, 90, 60*
Duff moisture	%	100, 50, 25*
Weather		
Midflame wind speed	mph	4
Terrain		
Slope grade	%	0

*Dry, very dry, and extremely dry moisture data estimated based on similar FCCS moisture scenarios and “Guidelines for estimating live fuel moisture content” (BehavePlus v.5.0.3 Missoula, MT).

throughout the study to provide a more representative sample of my study sites (Susan Prichard, personal communication). For instances in which no field data were available, the FCCS default parameters provided within Fuelbed No. 282 (i.e., loblolly pine forest with no change agent) were kept in place for the control and EPO sites, and parameters within Fuelbed No. 158 (i.e., loblolly pine-shortleaf pine-mixed hardwoods forest with insects and disease change agent) were kept in place for the LPO sites. Only those parameters customized with field data are described in detail below; any other input variables within FCCS, for which no field data were available, used the default parameters provided within the respective fuelbeds. For every instance in which field data were available, the field data values were used in substitution for the default fuelbed values.

Each fuelbed was categorized by six horizontal strata within FCCS: canopy, shrubs, non-woody vegetation, woody fuels, litter-lichen-moss, and ground fuels. Some of these strata were then further categorized into sub-strata. The canopy stratum contains three categories: trees, snags, and ladder fuels. Field data were used to determine relative cover per species in the live tree canopy overstory and understory; no values were used in midstory. Trees per acre for both live canopy fuels and dead snags were calculated by using basal area and diameter at breast height (DBH) values. Basal area was obtained from field measurements and DBH was obtained from FCCS default values.

In addition to snag density, percent of snags by decay class was also necessary for input within the snag category. Since no field data were collected for snag decay class, it was assumed that the decomposition of snags would vary between control, EPO, and

LPO stands. Four decay classes were included in the snag category within FCCS: Class 1a (with foliage), Class 1b (bark, branches, and tops intact but without foliage), Class 2 (shed fine branches but retain course branches) and Class 3 (contains heartwood decay and no bark or branches remain). Percent snags within control stands were distributed between Class 1 (without foliage) (20%), Class 2 (40%), and Class 3 (40%) snags, using the same percentages as the original FCCS parameters. Snag count collected from the field was distributed unequally between decay Class 1 (without foliage) (5%), Class 2 (90%), and Class 3 (5%) for EPO stands to account for the large number of standing snags observed in the field. LPO stands were assumed to contain only Class 3 snags (100%) since no branches or bark were observed on these snags in the field.

The shrub stratum describes primary and secondary layers of shrubs. The non-woody fuels stratum describes herbaceous vegetation including grasses, sedges, rushes, and forbs in primary and secondary layers. Field data were used as inputs for percent shrub cover and percent cover for the primary layer of non-woody fuels. The woody fuels stratum contains five categories which describe down and dead fuels: all woody, sound, rotten, stumps, and accumulations. The ‘all woody’ category describes the depth and percent cover of sound and rotten woody fuels. Field data were used as inputs for fuelbed depth. The ‘sound wood’ category records species and fuel loading data for 1-, 10-, 100-, and a range of 1000-hr size classes. The ‘rotten wood’ category only records species and fuel loading data for a range of 1000-hr size classes. Field data were used to determine all input values for both the ‘sound’ and ‘rotten’ categories. The ‘stump’ category describes stumps within three sub-categories (sound, rotten, and lightered pitchy). FCCS default parameters only include rotten and lightered pitchy stump data within Fuelbed No. 282; therefore ’sound’ input values were added to the EPO stands to account for recently fallen SPB-killed trees. The ‘accumulation’ category describes types of accumulation in three sub-categories (jackpots, piles, and windrows). ‘Piles’ and ‘windrows’ were kept as “not present” for all stands. ‘Jackpots’ were changed to “not present” for control stands, whereas FCCS default parameters were kept in place for post-outbreak stands to account for fallen SPB-killed trees that may have piled on one another.

The litter-lichen-moss stratum describes several aspects of litter, lichen, and moss layers within each fuelbed. Depth and percent relative cover of litter (per litter type) were obtained from field data. The ground fuels stratum contains three categories: duff, squirrel middens, and basal accumulations. Depth and percent cover of duff were obtained from field data; however, since no distinction was made between the O_e (upper duff) and O_a (lower duff) layers in the field, total duff depth as measured in the field was split into equal amounts for upper and lower duff inputs, akin to the example fuelbed shown in Riccardi (2007).

The ‘Environmental Variables’ screen allows environmental input parameters (i.e., wind speed, fuel moisture scenario, and slope gradient) to be customized to user specifications. Predicted fire behavior outputs are based on input environmental conditions; changes in environmental conditions will produce different results. For this study, midflame wind speed was kept at the FCCS default value of 4 mph (6.44 km/h). This windspeed is consistent with values in this region according to USDA Forest Service Southern Research Station staff and results from other fire studies conducted in similar loblolly pine-dominated stands within the CEF (Helen Mohr, personal communication; Mohr *et al.* 2004; Stottlemyer 2011). Custom moisture scenarios were created to reflect dry, very dry, and extremely dry fuel moisture conditions during the Piedmont fire season using the percentile conditions provided in Table 2. Crown and duff moisture values are included as part of the moisture scenario, however, since FireFamily Plus did not provide these output values, percentages were determined based similar FCCS default moisture scenarios. Percent crown moistures were given values of and 120, 90, 60 for dry, very dry, and extremely dry scenarios, respectively. Percent duff moistures were given values of 100, 50, and 25 for dry, very dry, and extremely dry scenarios, respectively. Similar to other published studies (Simard *et al.* 2011), the slope was held constant at zero percent for all sites to focus specifically on the effects of stand structure on fire behavior.

Once all input values were entered for each stand, FCCS was used to model surface fire behavior for each of the study sites. Flame length and rate of spread were derived as metrics of predicted surface fire behavior. Predicted fire behavior was then compared between stand types. Crosswalks to one of the 40 standard fuel models (Scott and Burgan 2005) under the set of environmental variables (fuel moisture, midflame windspeed, and slope gradient) that were specified were also obtained. The primary purpose of obtaining the crosswalks for this study was to provide the closest fuel model match in terms of predicted surface fire behavior for use and comparison within BehavePlus.

BehavePlus

Based on measured fuel data, fire behavior under various burning conditions was modeled using BehavePlus fire modeling system (v. 5.0.3 Missoula, MT). Similar to what was done in FCCS, custom fuel models were created on a stand basis within each study site, for a total of 16 control, 14 EPO, and 12 LPO custom fuel models. Simulations for all stands were developed using the dry, very dry, and extremely dry custom moisture data from the region. However, percentile 'Live Herbaceous Moisture' values obtained from FireFamily Plus were below the valid range permitted within BehavePlus (i.e., 30-300%); therefore, 30% (i.e., fully cured) was used for all sites as this was the lowest possible value that BehavePlus accepted. Fuel model types were constructed to be dynamic, meaning that 100% of live herbaceous fuel loads were transferred into the dead herbaceous fuel load since the live herbaceous moisture was considered to be fully cured. Under each modeled burning condition, differences in fire behavior were quantified and compared between stand types.

As was done within FCCS, identical midflame wind speeds (4 mph) were used when comparing stand types in order to remove the effect of canopy sheltering and directly compare the single influence of fuel on the fire behavior. Keeping the midflame windspeed consistent also allowed for a direct comparison between the two modeling systems using similar weather conditions. Similarly, as was done within the customized FCCS models and other studies of beetle-killed fuel complexes, the slope was kept at zero percent for all sites to focus specifically on the effects of stand structure on fire behavior.

Custom fuel models were initialized from existing standard fuel models (Scott and Burgan 2005) and were then modified based on fuels field data for each stand, where applicable. The custom fuel models were constructed using field-measured data from 1-, 10-, and 100-hr fuel loadings. For variables in which no field data were available, the default values provided within the respective standard fuel models were used. Table 3 depicts the parameters that were used for modeling within BehavePlus. Percent cover of live herbaceous and live woody fuels (i.e., shrubs) was recorded in the field; however, no field data were collected for live herbaceous or live woody fuel loadings. Therefore, fuel loading data in these categories were obtained from values provided within the output of the Strata and Categories Report of FCCS; a value of 0.017 tons/acre was applied to all live herbaceous fuel loads, whereas live woody fuel loads varied from stand to stand depending on percent woody shrub cover recorded during field data collection. All other fuel load values were obtained from converting field data to weights using equations given by Brown (1974) and were the same loading values as used within FCCS.

For control stands, the 'Long-needle Litter (TL8)' fuel model was used to initialize the parameters for customization based on the similarity of fuel loadings between this fuel model and the field data. The fuel loading values of the control stands were most similar to the input values provided within this model more so than any other of the standard fuel models compiled by Scott and Burgan (2005). This model overestimated 1-hr fuel loading and underestimated fuelbed depth, but appeared to give reasonable estimates of other components. The 'TL8' fuel model is described as not

Table 3. Parameters used in custom fuel models within BehavePlus for three Piedmont study sites.

Parameter	Units	Value
Fuel/Vegetation, Surface/Understory		
Fuel model type	N/A	dynamic
1-hr time lag fuel load	tons/ac	field/regression equation
10-hr time lag fuel load	tons/ac	field/regression equation
100-hr time lag fuel load	tons/ac	field/regression equation
Live herbaceous fuel load	tons/ac	0.017[1]
Live woody fuel load	tons/ac	field/FCCS output[1]
1-h SA/V	ft^2/ft^3	fuel model default
Live herbaceous SA/V	ft^2/ft^3	fuel model default
Live woody SA/V	ft^2/ft^3	fuel model default
Fuel bed depth	in	field/regression equation
Dead fuel moisture of extinction	%	fuel model default
Dead fuel heat content	Btu/lb	fuel model default
Live fuel heat content	Btu/lb	fuel model default
Fuel moisture		
1-hr time lag fuel moisture	%	percentile data[2]
10-hr time lag fuel moisture	%	percentile data[2]
100-hr time lag fuel moisture	%	percentile data[2]
Live herbaceous moisture	%	30[3]
Live woody moisture	%	percentile data[2]
Weather		
Midflame wind speed (upslope)	mph	4
Terrain		
Slope grade	%	0

[1]Obtained from FCCS output reports.
[2]Percentile moisture data as seen in Table 2.
[3]Minimum input value permitted by BehavePlus.

having been recently burned; containing dead and down woody fuel (litter) beneath a forest canopy; composed of long-needle pine litter; moderate load and compactness that may include small amounts of herbaceous load; moderate spread rate; and low flame length (Scott and Burgan 2005).

Although the slash-blowdown (SB) fuel models are described as being able to model forested areas with heavy mortality (Scott and Burgan 2005), loading values from 1-, 10-, and 100-hr fuels data collected from post-outbreak stands during this study did not closely match the loading values provided within this standard fuelbed. When comparing fuel loading values of the EPO stands to the fuel models compiled by Scott and Burgan (2005), it was found that the timber litter model 'Small Downed Logs (TL4)' had input values that were most appropriate. Although this model slightly underestimated biomass for 10-hr fuel loading and greatly underestimated fuelbed depth, estimates of other components appeared to be reasonable. Therefore, the 'TL4' fuel model was used to initialize parameters for the customized EPO fuel models. The primary carrier of fire for 'TL4' is described to be a moderate load of fine litter and coarse fuels, including small diameter downed logs (Scott and Burgan 2005).

When comparing fuel loading values of LPO stands to the fuel models described by Scott and Burgan (2005), it was found that the timber litter model 'Large Downed Logs (TL7)' had input values that were most appropriate, although this model slightly overestimated biomass for 100-hr fuel loading and greatly underestimated fuelbed depth. Estimates of other components appeared to be reasonable. Therefore, the 'TL7' fuel model was used to initialize parameters for the LPO stands. The primary carrier of fire for this model is described to be a heavy load of forest litter, including large diameter downed logs (Scott and Burgan 2005).

The predicted fire behavior of several standard fuel models (Scott and Burgan 2005) was compared to predicted fire behavior of customized fuel models in order to determine how closely the predicted behavior for the standard fuel model matched the custom models. Since standard fuel models 'TL8,' 'TL4,' and 'TL7' were used to initiate parameters within the control, EPO, and LPO stands, respectively, these models were used for comparison. Other studies modeling wildfire within SPB-killed stands have used the slash-blowdown fuel model 'Low Load Activity Fuel (SB1)' to model this fuel complex (Stottlemyer 2011). In addition, this fuel model is described for use in forested areas with heavy mortality. Therefore, fuel model 'SB1' was also used for comparison against post-outbreak stands.

Statistical Analyses

Fuel loading and stand conditions were analyzed with data collected from September 2009 to January 2010. Comparisons were made between control and post-outbreak stands for each study area. Comparisons were also made between the EPO and LPO stands within the SNF since both stand types existed on this study area. Differences in mean fuel loading (1-, 10-, 100- and 1000-hr fuels), litter depth, duff depth, fuelbed depth, basal area, percent woody and non-woody coverage, and slope among the chronosequence were examined using analysis of variance. For all analyses, tests of normality were investigated using probability plots and using the Shapiro-Wilk *W*-test

(Zar 1999; α = 0.05). If data were non-normal, they were transformed using log or square root functions, then reanalyzed. When analyses were significant, Tukey's HSD test (α = 0.05) was used to identify differences among means. Results produced by FCCS and BehavePlus were used for analyses of fire behavior. Pairwise comparisons were used to determine significant differences among surface rate of spread and flame length for control and post-outbreak stands within the CEF and the ONF, and an unpaired t-test was used for analysis of fire behavior within the SNF. For all analyses alpha was set at 0.05, although actual *p*-values are reported throughout. All analyses were conducted using SAS version 9.2 (SAS 2008).

CHAPTER FOUR

RESULTS

Downed Woody Fuel Dynamics

Measurements of downed woody fuels were collected from control and post-outbreak stands killed by SPB at different years. Mean downed woody fuel (DWF) loads and fuel dynamics were then characterized and compared between stand types within each study area. Results provide baseline measurements of fuel loading for SPB-killed forest stands at 2 and 8 years since outbreak, as well as a comparison of fuel dynamics between these stand types.

Control Stands (0 Years Since Outbreak)

Total DWF loading was lower in the control stands than in the post-outbreak stands for all sites. Fine fuel loads, which are the most related to surface fire behavior, were distributed unevenly in control stands among 1- (10.3%), 10- (36.6%), and 100-hr (53.1%) woody fuels. Total fine surface fuel loads were similar between study sites, averaging 3.34 tons/acre in the ONF, 3.18 tons/acre in SNF, and 3.45 tons/acre in the CEF (Tables 4-6). Mean coarse woody debris (CWD) (i.e., 1000-hr fuels) loading varied between study sites. CWD weights averaged 6.67 tons/acre in the ONF, 10.89 tons/acre in SNF, and 2.45 tons/acre in CEF. A majority of the CWD was classified as sound according to decay class as described by Pyle and Brown (1998).

Table 4. Comparison of mean fuel loading and surface fuel depth between control and early post-outbreak (EPO) stands within the Oconee National Forest (n=7 control and n=7 EPO) and the Sumter National Forest (n=3 control and n=7 EPO). Numbers in parentheses represent standard error. Means within a row followed by the same letter are not significantly different at the 0.05 level.

	Time Since Outbreak		
	0 Years (control)	2 Years (EPO)	*p*-value
Oconee National Forest			
1-hr Weight (t/ac)	0.39a (0.04)	0.86b (0.07)	0.0002
10-hr Weight (t/ac)	1.51a (0.27)	2.68b (0.27)	0.0031
100-hr Weight (t/ac)	1.44a (0.25)	4.64b (0.49)	0.0032
1000-hr Weight (t/ac)	6.67a (1.71)	31.94b (3.48)	0.0005
Litter Depth (in)	1.30a (0.15)	1.13a (0.09)	0.1029
Duff Depth (in)	1.09a (0.07)	1.03a (0.12)	0.3797
Dead Fuel Height (in)	9.82a (1.41)	18.41b (2.71)	0.0017
Sumter National Forest			
1-hr Weight (t/ac)	0.37a (0.02)	0.50a (0.05)	0.1244
10-hr Weight (t/ac)	0.91a (0.17)	1.60a (0.30)	0.1418
100-hr Weight (t/ac)	1.71a (0.49)	2.55a (0.56)	0.3930
1000-hr Weight (t/ac)	11.90a (3.22)	26.35a (3.95)	0.0818
Litter Depth (in)	1.34a (0.10)	1.57a (0.10)	0.2131
Duff Depth (in)	1.11a (0.07)	1.28a (0.08)	0.2374
Dead Fuel Height (in)	7.46a (2.31)	13.28a (2.54)	0.1696

Table 5. Comparison of mean fuel loading and surface fuel depth between early post-outbreak (EPO) (n=7) and late post-outbreak (LPO) (n=9) stands within the Sumter National Forest. Numbers in parentheses represent standard error. Means within a row followed by the same letter are not significantly different at the 0.05 level.

	Time Since Outbreak		
	2 Years (EPO)	8 Years (LPO)	*p*-value
Sumter National Forest			
1-hr Weight (t/ac)	0.50a (0.05)	0.40a (0.03)	0.0699
10-hr Weight (t/ac)	1.60a (0.30)	1.66a (0.19)	0.9067
100-hr Weight (t/ac)	2.55a (0.56)	3.85b (0.43)	0.0477
1000-hr Weight (t/ac)	26.35a (3.95)	25.67a (1.62)	0.9838
Litter Depth (in)	1.57a (0.10)	1.79a (0.10)	0.1670
Duff Depth (in)	1.28a (0.08)	1.47a (0.09)	0.1640
Dead Fuel Height (in)	13.28a (2.54)	13.59b (1.67)	0.5840

Table 6. Comparison of mean fuel loading and surface fuel depth between control and late post-outbreak (LPO) stands within the Clemson Experimental Forest (n=3 control and n=3 LPO) and Sumter National Forest (n=3 control and n=9 LPO). Numbers in parentheses represent standard error. Means within a row followed by the same letter are not significantly different at the 0.05 level.

	Time Since Outbreak		
	0 Years (control)	8 Years (LPO)	*p*-value
Clemson Experimental Forest			
1-hr Weight (t/ac)	0.27a (0.01)	0.41a (0.08)	0.2430
10-hr Weight (t/ac)	1.23a (0.13)	1.59a (0.28)	0.5238
100-hr Weight (t/ac)	1.95a (0.34)	6.73b (0.88)	0.0112
1000-hr Weight (t/ac)	2.45a (1.39)	25.44b (2.06)	<0.0001
Litter Depth (in)	2.13a (0.29)	2.20a (0.15)	0.8485
Duff Depth (in)	1.41a (0.33)	0.92a (0.10)	0.2273
Dead Fuel Height (in)	12.62a (1.45)	18.25b (1.37)	0.0047
Sumter National Forest			
1-hr Weight (t/ac)	0.33a (0.04)	0.40a (0.03)	0.3803
10-hr Weight (t/ac)	1.09a (0.63)	1.66a (0.19)	0.1362
100-hr Weight (t/ac)	1.95a (0.64)	3.85b (0.43)	0.0330
1000-hr Weight (t/ac)	9.58a (3.10)	25.67b (1.62)	0.0015
Litter Depth (in)	1.49a (0.01)	1.79a (0.10)	0.1409
Duff Depth (in)	1.31a (0.12)	1.47a (0.09)	0.4148
Dead Fuel Height (in)	6.55a (1.04)	13.59b (1.67)	0.0035

Mean diameter of downed stems (sound or rotten) for all study sites combined was 12.3 inches. Mean litter depth ranged from 1.30 inches in the ONF to 2.13 inches in the CEF. Mean duff depth ranged from 1.09 inches in the ONF to 1.41 inches in the CEF. Dead fuel height varied between study areas, ranging from 6.55 inches in the SNF to 12.62 inches in the CEF.

Early Post-Outbreak Stands (2 Years Since Outbreak)

Total fine surface fuel loads within EPO stands averaged 8.18 tons/acre in the ONF and 4.65 tons/acre in the SNF (Table 4). CWD loading varied between study sites; CWD weights averaged 31.94 tons/acre in the ONF and 26.35 tons/acre in the SNF. A majority of the CWD was classified as sound (Pyle and Brown 1998) for both the ONF (87%) and the SNF (69%). Mean diameter of downed stems (sound or rotten) was 15.7 inches for the ONF and 16.2 inches for the SNF. Mean litter depth varied between study sites, with 1.13 inches in the ONF and 1.57 inches in the SNF. Mean duff depth was 1.03 inches in the ONF and 1.28 inches in the SNF. Mean fuel height ranged from 13.28 inches in the SNF to 18.41 inches in the ONF.

Late Post-Outbreak Stands (8 Years Since Outbreak)

Total fine surface fuel loads averaged 8.73 tons/acre in the CEF and 5.91 tons/acre in the SNF (Table 6). CWD loading varied between sites. CWD weights averaged 25.44 tons/acre in the CEF and 25.67 tons/acre in the SNF. A majority of the CWD was classified as rotten according to decay class as described by Pyle and Brown

(1998) for both the CEF (51%) and the SNF (76%). Mean diameter of downed stems (sound or rotten) was 15.3 inches for the CEF and 16.2 inches for the SNF. Mean litter depth varied between study sites, with 2.20 inches in the CEF and 1.79 inches in the SNF. Mean duff depth was 0.92 inches in the CEF and 1.47 inches in the SNF. Mean fuel height ranged from 13.59 inches in the SNF to 18.25 inches in the CEF.

Comparison Between Stand Types

Downed woody fuel loads and surface fuel depth were compared between control stands and EPO stands within both the ONF and SNF (Table 4). Significant differences in DWF loads were found between stand types. Within the ONF, 1-, 10-, 100-, and 1000-hr fuels, as well as dead fuel height were significantly greater in EPO stands than in control stands (Table 4). Fuel loads and surface fuel depths were greater in EPO stands than in control stands for the SNF, but the difference was not significant. There were no significant differences in litter or duff depth between stand types for either of the study areas.

Fuel loading and surface fuel depths were also compared between EPO and LPO stands within the SNF (Table 5). Significant differences for 100-hr fuel loads were found between the stand types, with LPO stands containing greater fuel loads than EPO stands. No significant differences were detected for any other variables. Downed woody fuels were also compared between control stands and LPO stands within the CEF and SNF (Table 6). Significant differences in fuel loading were found between stand types. Within both the CEF and SNF, 100- and 1000-hr fuel loads, as well as dead fuel height were significantly greater in LPO stands than in control stands. There were no significant differences for 1- or 10-hr fuels, or for litter or duff depth between the control and LPO stands.

Stand Structure Dynamics

Stand characteristics were collected from control and post-outbreak stands killed by SPB at different years. Stand structure was compared between stands along the chronosequence (i.e., control, early post-outbreak, and late post-outbreak) within each of the three study sites (Tables 7-9). Photos depicting typical plots from each stand type (i.e., control, EPO, and LPO) are shown in Figure 4.

Control Stands (0 Years Since Outbreak)

Control stands contained the least amount of hardwood tree species in the overstory or understory as compared to post-outbreak stands. Loblolly pine was the dominant species (76.2%) of the living forest canopy within control stands. Basal area (BA)(ft^2/ac) of live conifers within the control stands ranged from 100.00 ft^2/acre (ONF) to 151.67 ft^2/acre (CEF). Dead trees located within control stands comprised a nominal portion of the total basal area and primarily consisted of loblolly pine. Percent shrub cover varied widely between study areas, ranging from 18.40% in the SNF to 44.61% in the CEF. Percent grass ranged from 0.42% in the CEF to 5.66% in the ONF. Percent forb cover ranged from 0.28% in the SNF to 1.22% in the ONF.

Table 7. Comparison of mean stand characteristics between control and early post-outbreak (EPO) stands within the Oconee National Forest (n=7 control and n=7 EPO) and the Sumter National Forest (n=3 control and n=7 EPO). Numbers in parentheses represent standard error. Means within a row followed by the same letter are not significantly different at the 0.05 level.

	Time Since Outbreak		
	0 Years (control)	2 Years (EPO)	*p*-value
Oconee National Forest			
BA Live Conifer*	100.00a (10.75)	2.14b (1.38)	0.0001
BA Dead Conifer	3.57a (2.36)	75.36b (6.58)	<0.0001
BA Live Hardwood	32.50a (7.40)	58.93a (9.43)	0.0652
BA Dead Hardwood	1.43a (1.43)	2.86a (2.47)	0.6335
Total BA	137.50a (12.13)	139.29a (12.59)	0.8952
Sapling Density (sap/ac)	1778.62a (319.31)	1918.42a (383.68)	0.5852
% Shrub Coverage	26.43a (4.20)	18.78a (1.82)	0.0808
% Grass Coverage	5.66a (1.70)	5.79a (1.99)	0.7716
% Forbs Coverage	1.22a (0.51)	0.57a (0.31)	0.1635
Sumter National Forest			
BA Live Conifer	141.67a (17.22)	12.86b (4.61)	<0.0001
BA Dead Conifer	10.00a (2.50)	84.29b (10.64)	0.0005
BA Live Hardwood	40.00a (18.76)	63.93a (11.48)	0.2946
BA Dead Hardwood	0.83a (0.83)	7.86a (4.02)	0.3032
Total BA	192.50a (13.77)	168.93a (8.07)	0.1592
Sapling Density (sap/ac)	1698.30a (382.90)	2149.79a (229.76)	0.3237
% Shrub Coverage	16.94a (5.78)	32.86a (6.98)	0.2138
% Grass Coverage	3.13a (1.27)	5.30a (0.90)	0.1488
% Forbs Coverage	1.11a (0.73)	1.34a (0.24)	0.7045

*BA= Basal Area (ft^2/ac)

Table 8. Comparison of mean stand characteristics between early post-outbreak (EPO) (n=7) and late post-outbreak (LPO) (n=9) stands within the Sumter National Forest. Numbers in parentheses represent standard error. Means within a row followed by the same letter are not significantly different at the 0.05 level.

	Time Since Outbreak		
	2 Years (EPO)	8 Years (LPO)	*p*-value
Sumter National Forest			
BA Live Conifer*	12.86a (4.61)	23.33a (6.26)	0.3025
BA Dead Conifer	84.29a (10.64)	18.06b (5.87)	<0.0001
BA Live Hardwood	63.93a(11.48)	79.44a (15.29)	0.4538
BA Dead Hardwood	7.86a (4.02)	1.11a (0.61)	0.0805
Total BA	168.93a (8.07)	121.94b (14.17)	0.0187
Sapling Density (sap/ac)	2149.79a (229.76)	2215.66a (163.76)	0.8137
% Shrub Coverage	32.86a (6.98)	37.06a (4.81)	0.5063
% Grass Coverage	5.42a (0.89)	7.36a (1.24)	0.4013
% Forbs Coverage	1.34a (0.24)	0.58b (0.24)	0.0326

*BA= Basal Area (ft^2/ac)

Table 9. Comparison of mean stand characteristics between control and late post-outbreak (LPO) stands within the Clemson Experimental Forest (n=3 control and n=3 LPO) and the Sumter National Forest (n=3 control and n=9 LPO). Numbers in parentheses represent standard error. Means within a row followed by the same letter are not significantly different at the 0.05 level.

	Time Since Outbreak		
	0 Years (control)	8 Years (LPO)	*p*-value
Clemson Experimental Forest			
BA Live Conifer*	151.67a (6.51)	1.67b (0.83)	0.0031
BA Dead Conifer	5.83a (5.83)	8.33a (0.83)	0.6014
BA Live Hardwood	47.50a (5.20)	69.17a (9.61)	0.2832
BA Dead Hardwood	2.50a (2.50)	0.00a (0.00)	0.4227
Total BA	207.50a (4.33)	79.17b (9.46)	0.0098
Sapling Density (sap/ac)	1495.85a (179.95)	2654.30a (148.78)	0.1400
% Shrub Coverage	44.61a (5.12)	45.90a (4.25)	0.8602
% Grass Coverage	0.42a (0.42)	12.29a (11.88)	0.4090
% Forbs Coverage	0.49a (0.30)	0.28a (0.28)	0.6388
Sumter National Forest			
BA Live Conifer	141.67a (15.30)	23.33b (6.26)	<0.0001
BA Dead Conifer	6.67a (4.41)	18.06a (5.87)	0.1769
BA Live Hardwood	34.17a (3.00)	79.44a (15.29)	0.1293
BA Dead Hardwood	3.33a (3.33)	1.11a (0.61)	0.3001
Total BA	185.83a (12.02)	121.94b (14.18)	0.0344
Sapling Density (sap/ac)	2069.45a (143.59)	2215.66a (163.76)	0.6389
% Shrub Coverage	18.40a (0.96)	37.06b (4.81)	0.0227
% Grass Coverage	1.11a (0.59)	7.36b (1.27)	0.0095
% Forbs Coverage	0.28a (0.13)	0.58a (0.24)	0.5051

*BA= Basal Area (ft^2/ac)

Figure 4. Photos of typical stand types measured within this study. From left to right: control, early post-outbreak, late post-outbreak. Photos were taken between September 2009 – January 2010.

Comparison Between Stand Types

Stand structure was compared between control and EPO stands (Table 7). Total basal area was similar between stand types within the ONF (137.50 ft^2/acre and 139.29 ft^2/acre for control and EPO stands, respectively); however, the composition of this basal area differed between stand types. For example, basal area of dead conifers comprised about half of the total basal area within each study area for EPO stands, whereas control stands primarily consisted of live pine. Within both the ONF and SNF, basal area of live conifers was significantly greater in the control stands than in the EPO stands. Alternatively, basal area of dead conifers was significantly greater in the EPO stands than in the control stands for both study sites. The basal area of standing dead hardwoods and percent forb cover were negligible in both study sites, and were not significantly different between stand types at the 0.05 level.

Stand structure was also compared between EPO and LPO stands within the SNF (Table 8). Significant differences were detected between stand types for dead conifers, total basal area, and percent forb coverage. All three variables were greater in the EPO stands than in the LPO stands. No significant differences between stand types were found for any other variables. Stand structure was also compared between control and LPO stands (Table 9). Basal area of live conifers as well as total basal area (live and dead conifers and hardwoods combined) was significantly higher in the control stands than in the LPO stands for both the CEF and SNF. A majority of the total basal area was comprised of live hardwood for both study sites (87% within the CEF and 65% within the SNF). Basal area of standing dead conifers within LPO stands were not significantly different from control stands, and were almost down to pre-outbreak levels (i.e., 8.33 ft^2/acre for post-outbreak and 5.83 ft^2/acre for control stands).

Percent woody vegetation (i.e., shrub) and grass cover were significantly greater in LPO stands than control stands within the SNF (Table 9). There were no significant differences in percent shrub, grass, or forb cover between control and LPO stands within the CEF. While sapling density was greater in LPO stands than in control stands for both study sites, it was not significant at the 0.05 level. Approximately 16% of saplings in control stands were loblolly pine, whereas the remainder were primarily comprised of hardwood species. Loblolly pine comprised 1.9% and 3.4 % of total saplings within EPO and LPO stands, respectively. The remainder consisted of a mixture of hardwood species.

While the amount of total basal area was similar between stand types (i.e., control vs. post-outbreak), their composition was vastly different. Most of the total basal area within the control stands was comprised of live loblolly pine, whereas EPO stands were primarily comprised of standing dead loblolly pine, and LPO stands were dominated by live hardwoods. Figure 5 depicts the proportion of total basal area for pine and hardwood species within each stand type for all study sites combined. Very few dead hardwoods were observed, but were included in the totals for both oak-hickory and other hardwood species. Species richness was similar between stand types, however species density varied between control and post-outbreak stands. Hardwood species increased in density from control to EPO and LPO stages, with sweetgum (*Liquidambar styraciflua)* comprising the largest proportion of SPB-killed stands.

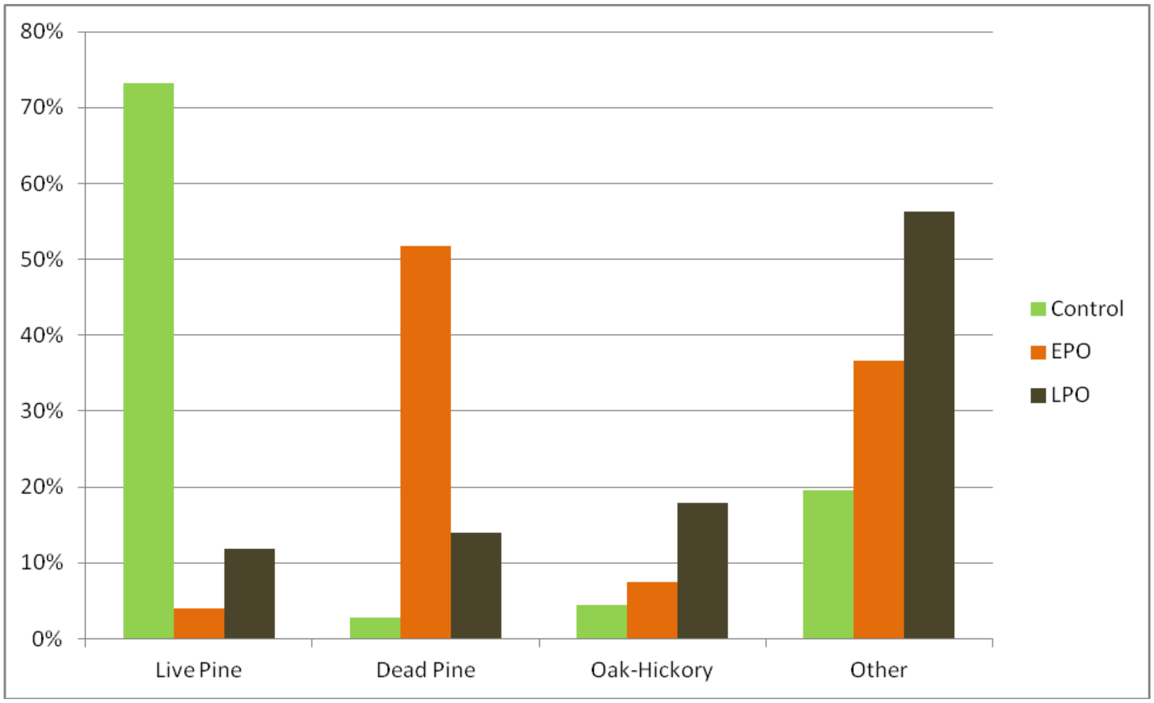

Figure 5. Proportion of total basal area by pine and hardwood species within control, early post-outbreak (EPO) and late post-outbreak (LPO) stands for all study sites combined.

Predicted Wildland Fire Behavior

Various moisture conditions were used during modeling to illustrate the changes in fire behavior that occur as conditions become drier. Since no fuel moisture data for the Piedmont were available in the published literature and were not collected in the field as a part of this study, fuel moisture conditions for 80th, 90th, and 99th percentiles were obtained from FireFamily Plus. The 80th, 90th, and 99th percentile fire weather represent dry, very dry, and extremely dry moisture conditions, respectively. Table 10 depicts the results produced by FireFamily Plus, which were subsequently used to create custom moisture scenarios within FCCS and BehavePlus.

Wildfire behavior characteristics (rate of spread (ROS) and flame length) were predicted for stands along the chronosequence within each of the study sites under each of the three varying moisture conditions. Results of wildfire behavior characteristics predicted by FCCS and BehavePlus are provided in Tables 11-13. Predictions from the two modeling systems vary slightly, with FCCS typically predicting faster ROS and higher flame lengths than BehavePlus. However, both systems showed the same pattern of fire behavior occurring with increasing drought conditions; increased behavior was predicted when moving from dry to extremely dry conditions, although specific outputs differed between stands.

Rate of Spread

Both FCCS and BehavePlus predicted the slowest ROS to be in the control stands, under dry moisture conditions for all study sites (Tables 11-13). Results provided by

Table 10. Fuel moisture values used to create custom moisture scenarios within FCCS and BehavePlus. These values were derived from historic weather analyses using the following Remote Access Weather Stations (RAWS): Shoal Creek (ID # 012902), Oconee #1 (ID# 093701), Whitmire (ID# 380902), and Uwharrie (Troy) (ID# 317001). Years 1965 – 2011 (November 1 – April 30) were included.

Parameter	80^{th} Percentile (Dry)	90^{th} Percentile (Very Dry)	99^{th} Percentile (Extreme)
1-hr time lag fuel moisture (%)	6	5	2
10-hr time lag fuel moisture (%)	8	7	6
100-hr time lag fuel moisture (%)	16	14	12
1000-hr time lag fuel moisture (%)	20	19	17
Herbaceous moisture (%)	27	13	5
Live woody moisture (%)	97	76	70

Table 11. Comparison of mean fire behavior (rate of spread (ft/min) and flame length (ft)) between control and early post-outbreak (EPO) stands within the Oconee National Forest (n=7 control and n=7 EPO) and within the Sumter National Forest (n=3 control and n=7 EPO) as predicted by FCCS and BehavePlus. Results were based on 80^{th}, 90^{th}, and 99^{th} percentile fuel moisture conditions and a midflame windspeed of 4mph. Means within a row for each modeling program followed by the same letter are not significantly different at the 0.05 level.

	FCCS			BehavePlus		
Fuel Moisture Condition	0 Years (control)	2 Years (EPO)	*p*-value	0 Years (control)	2 Years (EPO)	*p*-value
Oconee National Forest						
Rate of Spread						
Dry	5.5a	10.2b	0.0001	4.2a	8.3b	0.0144
Very Dry	6.1a	11.7b	<0.0001	5.3a	9.8b	0.0185
Extremely Dry	8.1a	15.1b	0.0001	6.4a	12.3b	0.0165
Flame Length						
Dry	3.4a	4.5b	0.0034	2.0a	3.2b	0.0003
Very Dry	3.6a	4.9b	0.0026	2.3a	3.5b	0.0002
Extremely Dry	4.5a	5.9b	0.0044	2.6a	4.2b	0.0003
Sumter National Forest						
Rate of Spread						
Dry	5.3a	8.1b	0.0095	3.4a	5.2a	0.1768
Very Dry	5.9a	9.4b	0.0077	4.3a	6.4a	0.1876
Extremely Dry	7.9a	11.9b	0.0142	5.2a	7.9a	0.1826
Flame Length						
Dry	3.1a	3.4a	0.2537	1.5a	2.2a	0.0942
Very Dry	3.3a	3.8a	0.2011	1.7a	2.5a	0.0934
Extremely Dry	4.2a	4.5a	0.2912	2.0a	2.9a	0.0922

Table 12. Comparison of mean fire behavior (rate of spread (ft/min) and flame length (ft)) for 16 (n=7 early post-outbreak (EPO) and n=9 late post-outbreak (LPO)) stands within the Sumter National Forest as predicted by FCCS and BehavePlus. Results were based on 80th, 90th, and 99th percentile fuel moisture conditions and a midflame windspeed of 4mph. Means within a row for each modeling program followed by the same letter are not significantly different at the 0.05 level.

	FCCS			BehavePlus		
Fuel Moisture Condition	2 Years (EPO)	8 Years (LPO)	*p*-value	2 Years (EPO)	8 Years (LPO)	*p*-value
Sumter National Forest						
Rate of Spread						
Dry	8.1a	10.8b	0.0094	5.2a	3.9a	0.1446
Very Dry	9.4a	13.4b	0.0020	6.4a	4.9a	0.1573
Extremely Dry	11.9a	17.3b	0.0012	7.9a	5.8a	0.1308
Flame Length						
Dry	3.4a	4.1a	0.0570	2.2a	2.3a	0.3778
Very Dry	3.8a	4.6b	0.0257	2.5a	2.6a	0.3917
Extremely Dry	4.5a	5.7b	0.0314	2.9a	3.0a	0.4803

Table 13. Comparison of mean fire behavior (rate of spread (ft/min) and flame length (ft)) between control and late post-outbreak (LPO) stands within the Clemson Experimental Forest (n=3 control and n=3 LPO) and within the Sumter National Forest (n=3 control and n=9 LPO) as predicted by FCCS and BehavePlus. Results were based on 80th, 90th, and 99th percentile fuel moisture conditions and a midflame windspeed of 4mph. Means within a row for each modeling program followed by the same letter are not significantly different at the 0.05 level.

	FCCS			Behave Plus		
Fuel Moisture Conditions	0 Years (control)	8 Years (LPO)	*p*-value	0 Years (control)	8 Years (LPO)	*p*-value
Clemson Experimental Forest						
Rate of Spread						
Dry	6.3a	10.5a	0.1043	4.1a	4.9a	0.1874
Very Dry	6.9a	13.1a	0.0686	5.1a	6.1a	0.1915
Extremely Dry	9.4a	16.8a	0.0685	6.0a	7.1a	0.1928
Flame Length						
Dry	3.4a	4.1a	0.1483	2.1a	3.2a	0.0524
Very Dry	3.6a	4.7a	0.0815	2.4a	3.6b	0.0486
Extremely Dry	4.5a	5.7a	0.0833	2.7a	4.0b	0.0457
Sumter National Forest						
Rate of Spread						
Dry	5.5a	10.8b	<0.0001	2.5a	3.9a	0.1209
Very Dry	6.0a	13.4b	<0.0001	3.1a	4.9a	0.1169
Extremely Dry	8.2a	17.3b	<0.0001	3.7a	5.8a	0.1288
Flame Length						
Dry	3.3a	4.1a	0.0706	1.4a	2.3b	0.0130
Very Dry	3.5a	4.6b	0.0332	1.6a	2.6b	0.0137
Extremely Dry	4.4a	5.7b	0.0369	1.8a	3.0b	0.0149

FCCS predicted the fastest ROS to be in the LPO stands under extremely dry conditions. Alternatively, BehavePlus predicted the fastest ROS to be in the EPO stands under extremely dry conditions. FCCS predicted ROS to be somewhat similar between all EPO and LPO stands under dry moisture conditions, but ROS became faster in the LPO than in EPO stands as moisture conditions became drier.

There were differences in predicted fire behavior between control and EPO stands (Table 11). Both FCCS and BehavePlus predicted significantly faster ROS in the EPO stands than in control stands for all moisture scenarios within both the ONF and SNF. ROS increased with increasing drought conditions. Predicted ROS by FCCS within control stands ranged from 5.3 ft/min within the SNF under dry conditions to 8.1 ft/min within the ONF under extremely dry conditions. Predicted ROS by FCCS within EPO stands ranged from 8.1 ft/min within the SNF under dry conditions to 15.1 ft/min within the ONF under extremely dry conditions.

Fire behavior was compared between EPO and LPO stands within the SNF (Table 12). FCCS predicted significantly higher ROS in LPO stands than in EPO stands for all moisture conditions. BehavePlus did not predict any significant differences between stand types. There were also significant differences in predicted fire behavior between control and LPO stands (Table 13). Within the CEF, ROS was faster in the LPO stands than in control stands, but was not significantly different at the 5% level in any of the moisture scenarios as predicted by FCCS or BehavePlus. FCCS predicted significantly faster ROS in LPO stands than in control stands for all percentile moisture conditions within the SNF. BehavePlus did not predict any significant differences in ROS for the

SNF. Predicted ROS by FCCS within control stands ranged from 5.5 ft/min within the SNF under dry conditions to 9.4 ft/min within the CEF under extremely dry conditions. Predicted ROS by FCCS within LPO stands ranged from 10.5 ft/min within the CEF under dry conditions to 17.3 ft/min within the SNF under extremely dry conditions.

Flame Length

Flame lengths were compared between stand types (Tables 11-13). Results provided by FCCS predicted the highest flame lengths (i.e., 5.9 ft) to be in EPO stands within the ONF at extreme moisture conditions; the lowest flame lengths (i.e., 3.1 ft) were predicted in control stands at dry moisture conditions within the SNF. BehavePlus made similar predictions, although the output values were lower than those of FCCS.

There were significant differences in flame length between control and EPO stands (Table 11), as well as between control and LPO stands (Table 13). Within the ONF, flame length was significantly greater in EPO stands than in control stands for all percentile moisture conditions as predicted by both FCCS and BehavePlus. Within the CEF, flame lengths became significantly higher in LPO stands than in control stands under very dry and extremely dry moisture scenarios as predicted by BehavePlus; results were not significant under dry conditions. Predictions made by FCCS for the CEF indicated that flame lengths were higher in the LPO stands than in control stands, but were not significantly different at the 5% level. Within the SNF, BehavePlus predicted flame lengths to be significantly higher in LPO stands than in control stands under all moisture conditions. Flame lengths became significantly higher in LPO stands than in control stands only under very dry and extremely dry moisture scenarios as predicted by FCCS.

Customized Fuelbed and Standard Fuel Model Comparison

Associated crosswalks from the customized fuelbeds to the standard 40 fuel models (Scott and Burgan 2005) as predicted by FCCS are provided in Tables 14-15. The primary purpose of obtaining the crosswalks was to determine the closest fuel model match in terms of predicted surface fire behavior for subsequent use within BehavePlus. Crosswalks to fuel models varied between control and post-outbreak stands in all study sites, as well as between dry to extremely dry moisture conditions within a study site. As stated in Chapter Three, fuel loading characteristics of the measured control stands were most similar to fuel loadings of fuel model 'TL8' (long needle litter) as described by Scott and Burgan (2005); however, when fuelbeds for control stands were customized within FCCS, fuel model 'TL9' (very high load broadleaf litter) was the most commonly predicted fuel model based on predicted fire behavior, especially under dry to very dry conditions (Tables 14-15). Although it varied by stand, fuel model 'TU4' (dwarf conifer with understory) was the most commonly predicted fuel model for control stands under extremely dry conditions.

Crosswalk predictions for post-outbreak stands fluctuated between various timber understory (TU), shrub (SH), and slash-blowdown (SB) fuel models (Tables 14-15). A majority of the EPO stands were predicted to have the most similar fire behavior to fuel

Table 14. Associated crosswalks to standard fuel models (Scott and Burgan 2005) under varying moisture conditions for control and early post-outbreak (EPO) stands as predicted by FCCS. Results were based on 80th, 90th, and 99th percentile fuel moisture conditions and a midflame windspeed of 4 mph.

	Dry		Very Dry		Extremely Dry	
Fuel Moisture Condition	Control	EPO	Control	EPO	Control	EPO
Oconee National Forest						
Stand 1	TU2	TU2	TU2	TU4	TU3	TU3
Stand 2	TL9	TU4	TL9	TU4	TU5	TU3
Stand 3	TL9	TU4	TL9	TU3	TU2	TU3
Stand 4	TL9	TU3	TL9	TU3	TU2	SB3
Stand 5	TL8	SH6	TL9	TU4	TL9	TU3
Stand 6	TL9	TU4	TL9	TU3	TU4	TU3
Stand 7	TL9	TU3	SH6	SB2	TU3	SB3
Sumter National Forest						
Stand 1	TL9	TU4	TL9	SH6	TU3	TU3
Stand 2	TL9	TU2	TL9	TU2	SH6	SB2
Stand 3	TL8	TL9	TL9	TU4	TL9	TU4
Stand 4	n/a	TL9	n/a	TL9	n/a	SH6
Stand 5	n/a	TU4	n/a	SH8	n/a	TU4
Stand 6	n/a	TU3	n/a	TU3	n/a	SB3
Stand 7	n/a	TU4	n/a	SH6	n/a	TU4

Table 15. Associated crosswalks to standard fuel models (Scott and Burgan 2005) under varying moisture conditions for control and late post-outbreak (LPO) stands as predicted by FCCS. Results were based on 80th, 90th, and 99th percentile fuel moisture conditions and a midflame windspeed of 4 mph.

	Dry		Very Dry		Extremely Dry	
Fuel Moisture Condition	Control	LPO	Control	LPO	Control	LPO
Clemson Experimental Forest						
Stand 1	TL9	TU3	TL9	TU3	TU4	SB3
Stand 2	TL9	TL9	TL9	SH6	TU4	TU3
Stand 3	TL9	SH2	TL9	SH4	TU4	TU5
Sumter National Forest						
Stand 1	TL9	TU4	TL9	TU3	TU4	SB3
Stand 2	TL9	TL6	TL9	TL9	TU4	TU4
Stand 3	TL9	SH2	TL9	SH2	SH6	SH4
Stand 4	n/a	TU3	n/a	TU3	n/a	SB3
Stand 5	n/a	TU4	n/a	TU3	n/a	SB2
Stand 6	n/a	TU2	n/a	TU4	n/a	TU3
Stand 7	n/a	SH8	n/a	SH2	n/a	SH4
Stand 8	n/a	TU3	n/a	TU3	n/a	SH7
Stand 9	n/a	TU4	n/a	SH2	n/a	SH4

model ‘TU4’ in dry and very dry conditions, and ‘TU3’ (i.e., moderate load, humid climate timber-grass-shrub) in extremely dry conditions (Table 14). Fuel model crosswalks varied widely between LPO stands, but a majority were predicted to have the most similar fire behavior to ‘TU3’ in dry and very dry conditions, and ‘SB3’ (i.e., high load activity fuel or moderate load blowdown) and ‘SH4’ (i.e., low load, humid climate timber-shrub) in extreme conditions (Table 15).

Rate of spread and flame length of standard fuel models were compared to those of customized models to determine which standard fuel model’s predicted fire behavior most closely relates to the SPB-killed fuel complex (Figures 6-7). The results of the standard fuel models were obtained under dry conditions (D2L2 moisture scenario) using BehavePlus. These particular fuel models were chosen for comparison from the standard 40 (Scott and Burgan 2005) based either on crosswalk predictions made by FCCS or based on similar fuel loads as previously described. Standard fuel models ‘TL8,’ ‘TL9,’ and ‘TU4’ were compared to control stands; standard fuel models ‘TU4,’ ‘TU3,’ ‘SB1,’ and ‘TL4’ were compared to EPO stands; and standard fuel models ‘SB1,’ ‘TU3,’ ‘SB3,’ ‘SH4,’ and ‘TL7’ were compared to LPO stands.

Based on the results, fuel models ‘TL8,’ ‘TL9,’ and ‘SB1’ appear to be good predictors of ROS for control stands under dry and very dry conditions. Fuel model ‘TL9’ appears to be the closest predictor under dry conditions for EPO stands, while fuel model ‘TU4’ is the closest predictor of EPO stands under very dry conditions; no fuel models closely match ROS under extremely dry conditions. Fuel model ‘TU4’ provides the closest match for LPO stands under dry conditions; there do not appear to be any

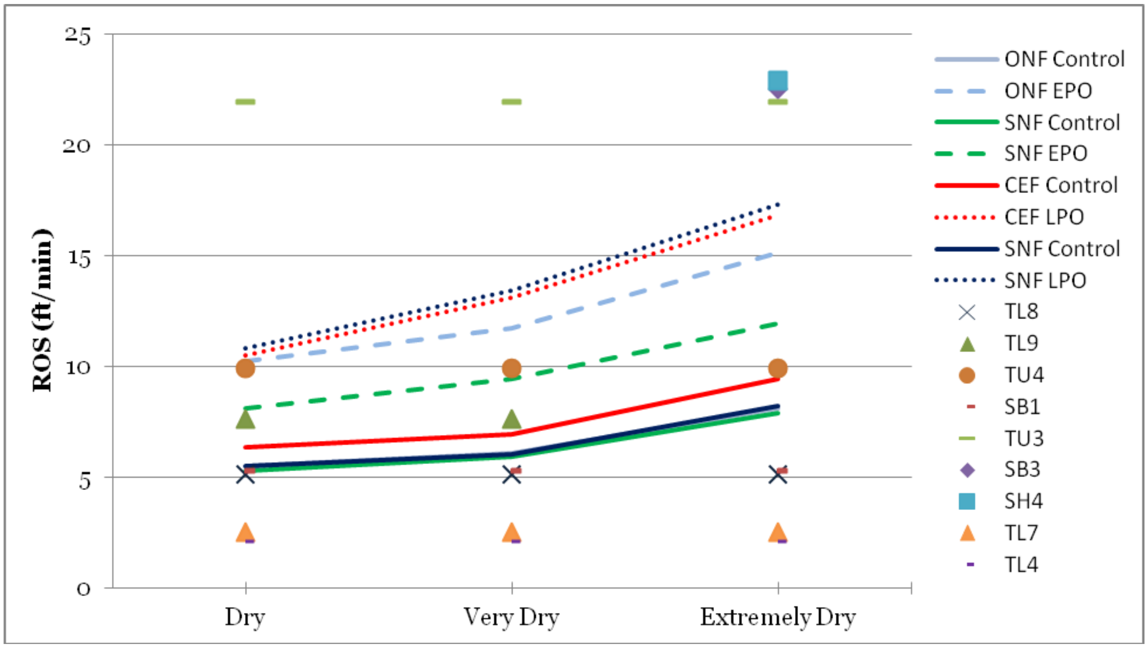

Figure 6. Comparison of mean rate of spread (ROS) between existing standard fuel models (Scott and Burgan 2005) and customized fuel models for control, early post-outbreak (EPE), and late post-outbreak (LPO) stands under varying moisture conditions. The standard fuel model results were obtained under dry conditions (D2L2 moisture scenario) using BehavePlus.

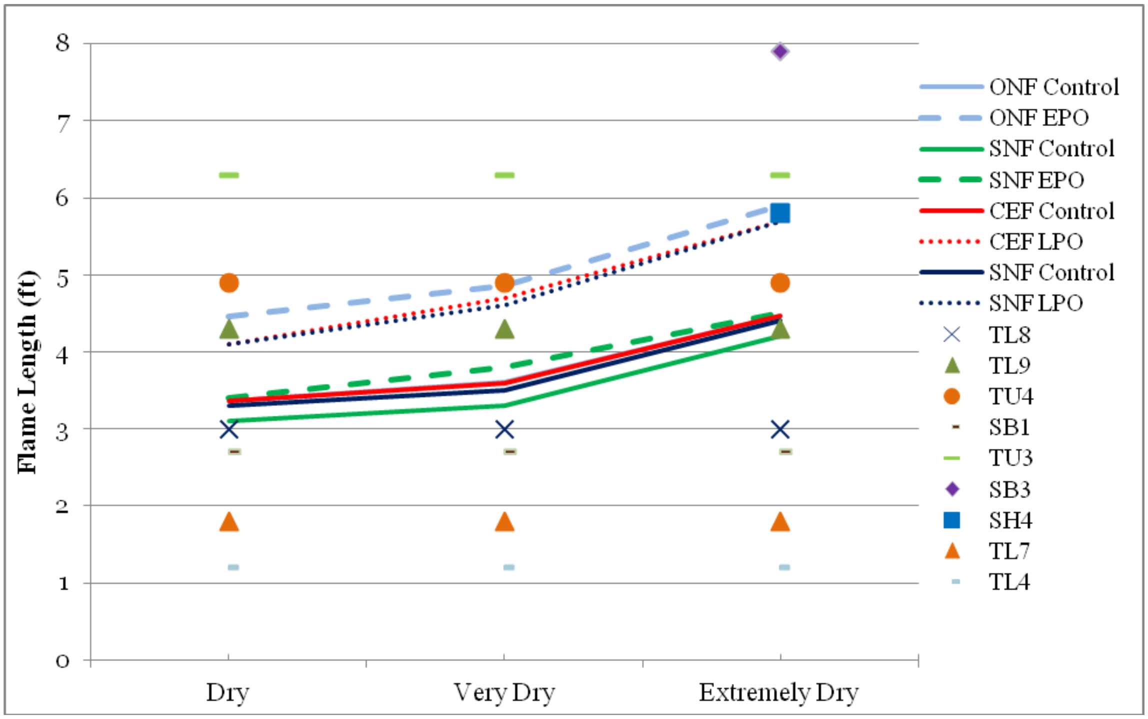

Figure 7. Comparison of mean flame length between existing standard fuel models (Scott and Burgan 2005) and customized fuel models for control, early post-outbreak (EPO), and late post-outbreak (LPO) stands under varying moisture conditions. The standard fuel model results were obtained under dry conditions (D2L2 moisture scenario) using BehavePlus.

standard fuel models that closely match ROS within LPO stands under very dry or extremely dry conditions. Similarly, when examining the relationship between standard fuel models and custom fuelbeds for flame length, fuel model ‘TL8’ appears to be the closest predictor for control stands under dry conditions, and fuel model TL9 provides the closest match under extremely dry conditions. Fuel models ‘TL9,’ ‘TU4,’ and ‘SH4’ are the closest predictors of flame length under dry, very dry, and extremely dry conditions, respectively, for the EPO stands within the ONF as well as the LPO stands within the CEF and SNF. No fuel models closely match flame length for EPO stands within the SNF.

CHAPTER FIVE

DISCUSSION

Several studies conducted in coniferous forest systems within the western U.S. have explored the relationship between beetle-caused changes in fuels and fire behavior (Brown 1975; Geiszler *et al.* 1980; Lotan *et al*. 1985; Lynch *et al.* 2006; Page and Jenkins 2007*b*; Romme *et al.* 1986*b*; Schoennagel *et al.* 2012; Simard *et al.* 2011). However, this is the first study to include fuels and fire behavior data in space (n=42 stands within n=3 study areas) and time (0-8 years since outbreak) within SPB-affected forest stands of the eastern U.S. Results from this study show how downed woody fuels, stand dynamics, and fire behavior within these stands change over time after an outbreak.

Downed Woody Fuel Dynamics in SPB-Killed Forest Stands

Based on the results of this study, a majority of the dead loblolly pines, or portions thereof, were still standing 2 years after an SPB outbreak; however, all needles had fallen to the ground by this time. It appears that 1- and 10-hr fuels continue to accumulate up to 2 years after an outbreak as small twigs and branches fall from the dead trees. By 8 years post-outbreak, fine fuel inputs have slowed since most of the smaller branches have already fallen to the forest floor. Fine fuel inputs and decomposition appear to reach a balance by this time, as there were no significant differences in fuel loading between control and LPO stands for 1- or 10-hr fuels. After most of the smaller branches have fallen, larger branches begin to break off, increasing 100-hr fuel loads on the forest floor. It appears that these 100-hr fuels continue to accumulate up to 8 years after outbreak as large branches continue to fall from dead trees, or fall with the snags if still attached to the trunk. After most branches have fallen from the snags, the snags begin to break off into sections or fall over whole. CWD loads appear to continue to accumulate up to 8 years after outbreak, although inputs slow by this time since a majority of the snags have fallen. Eight years post-outbreak does not appear to be long enough for CWD inputs and decomposition to reach a balance. However, it is anticipated that CWD loads will begin to decrease after this time as the fallen pines continue to decompose.

Comparison Between Control and EPO Stands

Fuel loading varied between control and EPO stands. A majority of the fuel loading for the ONF was significantly greater in the EPO stands than in the control stands, indicating that both small and large fuel inputs were still occurring at a rate that was more than offsetting the decomposition. The lack of significant differences in fuel loads between stand types within the SNF may be because many of the dead pines killed by SPB were still standing and had not yet acted as a significant input to loading. As the snags begin to break apart and fall over time, it is anticipated that fuel loads will become significantly greater in this study area as well. In addition, there was a high level of variability between stands in the SNF, which may have affected the results. Since the ONF and SNF have similar percentages of standing snags in their total basal area (i.e., 54% and 50%, respectively), it is possible that the ONF had a higher stand density than the SNF prior to outbreak, resulting in a larger input of downed woody fuels. The length of time that a snag remains standing depends on a variety of site and environmental factors and so it is unclear at what rate the remainder of these standing trees will continue to produce fine and CWD inputs. However, based on the regression model created by Radke *et al.* (2009) for rates of standing snag longevity, it can be expected that approximately 57% of an individual pine stem was down at the time of sampling within EPO sites. Therefore, it is anticipated that fine fuel inputs had slowed or stopped by this time since a majority of the branches had already fallen, and CWD inputs will continue to increase over the next few years as remaining trees begin to fall.

Significant differences in needle litter amount and depth were found between the MPB endemic and epidemic phases in western forests (Jenkins 2011; Klutsch *et al.* 2009; Page and Jenkins 2007*a*). Although no current outbreak stands were measured as a part of this study, it is expected that similar results would have been found as the needles began to fall to the forest floor. It has been indicated that litter accumulation and decomposition achieve a balance as litter decomposes to duff within one to two years (Jenkins 2011), which may explain the lack of significant differences in litter depth between control and EPO stands in this study.

Comparison Between Control and LPO Stands

Fuel loading also varied between control and LPO stands. Large (i.e., 100- and 1000-hr) downed woody fuels were significantly greater in LPO stands than in control stands for both the CEF and SNF, similar to findings in other field studies conducted within MPB-killed stands (Jenkins 2011; Klutsch *et al.* 2009; Page and Jenkins 2007*a*;

Simard *et al.* 2011). This difference in fuel load is likely due to a majority of the large branches from standing snags and/or the snags themselves having fallen to the forest floor by this time. Total fuel loads were approximately seven times higher and three times higher in LPO stands than in control stands for the CEF and SNF, respectively, which is similar to results by Nicholas and White (1984) who found that wood debris weights were four times higher in SPB-killed stands than in unaffected stands. If a fire were to burn these LPO stands, CWD fuel loading would likely further increase as the remaining dead snags fall after being burned. Stottlemyer (2011) found this to be the case when conducting a prescribed burn during spring months within SPB-killed stands in the CEF.

These differences in CWD fuel loads may indicate that decomposition and accumulation are not yet balanced at 8 years post outbreak. Smaller (i.e., 1- and 10-hr) fuel loads were not significantly different between stand types, indicating that inputs had slowed, and decomposition and accumulation had balanced by this time. Page and Jenkins (2007*a*) found that small fuels within MPB post-epidemic stands decay sufficiently over time so that the differences return to original levels at least after 20 years. The difference in time between results of this study (i.e., by 8 years) and Page and Jenkins (2007*a*) (i.e., 20 years) may be attributed to the faster decay process found in eastern forest systems as compared to western forests.

Eight years after an SPB outbreak in loblolly stands within the Piedmont appears to be enough time for stands to reach a balance between input and decomposition of fine fuels, but not for CWD. Based on findings by Jenkins (2011), it is expected that this balance will eventually be reached over time after all of the snags have fallen and the CWD decomposes. Radke *et al.* (2009) found that decomposition of residual loblolly CWD was nearly complete after 25 years. It is anticipated that it would take approximately the same length of time for loblolly CWD to decompose on these sites as well since a majority of the fallen trees had already begun to rot within the LPO stands.

In addition to fuel loading, fuelbed depth was also significantly greater in LPO stands than control stands. This is similar to results from studies conducted in other beetle-killed forests (Jorgansen and Jenkins unpublished; Page and Jenkins 2007*a*). Fuelbed depths were compared to those recorded by Stottlemyer (2011) on SPB-killed stands within the CEF that had been dead for 4-5 years; assuming site conditions were similar, it was found that fuelbed heights at 8 years after outbreak measured in this study were greater than in Stottlemyer's stands at 4-5 years post-outbreak. This is likely due to recently fallen snags piling upon each other or leaning up on other trees after falling.

Stand Structure Dynamics in SPB-Killed Forest Stands

As downed woody fuels change over time, stand structure changes simultaneously. As snags fall, the gaps they leave behind allow hardwoods and other early successional species to grow. If left untreated, the trajectory of forest stand composition after a beetle outbreak could transform from pine-dominated to hardwood-dominated forests. SPB outbreaks create gaps in the forest canopy that allow for increased sunlight and nutrient availability. However, the mineral soil that improves loblolly pine germination has limited exposure, and the canopy from surviving trees and/or new growth may both inhibit loblolly pine recruitment and favor the growth and establishment of more shade-tolerant species.

<u>Comparison Between Stand Types</u>

Similar to findings by Smith (1991), the post-outbreak stands that were previously loblolly-dominated became occupied by oaks and other low-quality hardwoods. It is assumed that a majority of the hardwood species observed in the canopy stratum of the post-outbreak stands were already established on the study sites at the time of initial SPB infestation; however the observed increase in basal area of live hardwoods from control to post-outbreak stands was likely due to growth allowed by the increased sunlight, water, and nutrient availability created by gaps. Similarly to the increase in hardwood growth observed here, Romme *et al.* (1986*b*) found that surviving trees grew more rapidly after MPB outbreak. Basal area of live conifers within the post-outbreak stands of the SNF was much greater than those of the ONF or CEF, indicating that outbreaks may not have been as severe within the SNF as they were in the other study areas and/or site or environmental conditions within the SNF may have been more conducive towards loblolly pine reestablishment, although it is unclear why this may be.

Percent shrub and grass cover were significantly greater in the LPO stands than in control stands for the SNF, indicating that re-growth of understory vegetation had increased by this time, possibly due to increases in light and nutrient availability caused by gaps. The results of this study provide insight into the stand dynamics associated with future stand composition resulting after SPB outbreak. Greater understanding of forest regeneration and succession in untreated SPB-killed forests will assist land managers to project forest recovery after an outbreak.

Predicted Wildland Fire Behavior in SPB-Killed Forest Stands

Given the large extent of Piedmont forests that have been affected by SPB outbreak and wildland fire in recent years, it has become increasingly important to understand expected fire behavior in these areas. Currently, prescribed burning is rarely used as a fuel reduction technique in SPB-killed areas because high intensities are expected, and fires could possibly damage soils, nearby trees, or target vegetation (Waldrop 2005). Stottlemyer *et al.* (2012) found that prescribed burning in beetle-killed areas may result in high-intensity fires, but if conducted properly and because of their limited area, can be controlled.

Fire type within the Piedmont is primarily surface fire; FCCS characterizes and summarizes variables that are especially relevant to surface fire behavior predictions, including surface fuel loading, arrangement, fuel morphology, species phenology (i.e., live shrub and nonwoody fuel by species), and ground coverage. Based on the results of this study, FCCS commonly predicted faster ROS and longer flame lengths than BehavePlus for all stand types. Empirical evidence of fire behavior in SPB-killed stands is not available, therefore it is unknown how closely predictions from either fire prediction modeling system represent actual fire behavior. However, in a prescribed fire conducted in the CEF 4-5 years after an SPB outbreak, it was observed that rates of spread were low to moderate, with relatively long residence times where piles of fuel were ignited. Flame lengths generally ranged from 1.5 to 13 feet with occasional torching of live trees (mid-story conifers, in particular) (Stottlemyer, personal communication). These observations were similar to predictions made by FCCS and BehavePlus for post-outbreak stands. Quantitative measurements of fire behavior during wildland fire situations (wildfire or prescribed fire) in SPB-killed areas are necessary to determine which fire prediction modeling system produces the most accurate results for the beetle-kill complex.

Similar to findings by Jenkins *et al.* (2008), post-outbreak stands in this study were found to have a substantial change in species composition and a highly altered fuel complex. In return, fire modeling systems using this altered fuel complex predicted different fire behavior than unaffected stands. Due to the abundance of fine fuels on the forest floor in stands within the EPO stage, particularly for the ONF, surface fire behavior was expected to increase as compared to the control stands. This was true with both FCCS and BehavePlus predictions, similar to findings in other studies (Jorgansen and Jenkins unpublished; Page and Jenkins 2007*b*; Simard *et al*. 2011). Flame length was predicted to be greatest in EPO stands within the ONF and in LPO stands under extremely dry conditions, likely due to the increased fuelbed height in these stands. ROS was predicted to be fastest in LPO stands by FCCS and in EPO stands by BehavePlus. ROS was predicted to be slowest in control stands for both modeling systems; this is likely because these stands contained the least amount of 1- and 10-hr fuels, which are primarily what drives the surface fire calculation. It would have been desirable to measure fuels and stand structure in stands beyond 8 years post-outbreak, however, no older stands were available because they had already been managed with salvage logging or other silvicultural activities. It is anticipated that elevated fire behavior may persist beyond 8 years post-outbreak, until fuel loads have decomposed enough to reach pre-outbreak levels.

Although not accounted for in this study in order to keep identical environmental conditions so that only the effect of fuels was being compared, the effect of wind is also expected to affect fire behavior. Some recent studies (Jenkins 2011; Page and Jenkins 2007*b*; Schoennagel *et al.* 2012) have suggested that there is an increased effect of wind in late post-outbreak stands due to the more open canopy than in sheltered, non-affected stands. Assuming this is correct, it is anticipated that fire behavior will increase even further in LPO stands than is shown in this study. In addition, it has also been assumed that due to the more open canopy in gray stage beetle-killed stands, surface fuel moistures would be slightly lower in these areas (Page and Jenkins 2007*b*; Schoennagel 2012). The results of my study show that fire behavior reacts differently to varying fuel moisture conditions. As expected, both rate of spread and flame length were predicted to be lower under dry moisture conditions than in extremely dry conditions. It should be noted that fuel moisture values provided by FireFamily Plus were used as inputs to customized moisture scenarios, and may have compounded the potential error associated with using these modeling systems.

Since great variability existed in the standard fuel model crosswalks predicted by FCCS for post-outbreak stands (i.e., timber understory, shrub, and slash-blowdown fuel models) (Tables 14-15), it appears that the standard fuel models currently being used in several fire modeling systems may not be good indicators of the SPB-killed fuel complex. Customized fuel models for SPB-killed areas should be used in order to obtain the most accurate results, although it is unclear as to how accurate the predictions are to actual fire behavior without having validation from fire behavior data. Although custom fuel models allow users to tailor input values depicting site specific fuel loadings and environmental conditions, advances in fire behavior prediction systems and/or creation of new standard fuel models may be necessary in order to accurately predict fire behavior in beetle-killed fuel complexes. Two systems (FIRETEC (Linn *et al.* 2002) and Wildland Fire Dynamics Simulator (Mell *et al.* 2007)) have recently been used by USDA Forest Service researchers (Russ Parsons, personal communication) to examine the beetle-killed fuel complex and may be useful in future research.

CHAPTER SIX

CONCLUSIONS AND MANAGEMENT IMPLICATIONS

Southern pine beetles play an integral role in coniferous forests of the Piedmont, but become extremely damaging at outbreak levels by killing valuable timber trees, generating dangerous and difficult working conditions, creating obstacles to forest management, and increasing fuel loading. It has been presumed that the increased surface and canopy dead fuels created by SPB outbreak increase the probability of extreme fire behavior, and subsequently increase the difficulty of fire control. Objectives of land managers have often included reducing fuel loads in these areas because of this presumption, however, supportive empirical data were previously lacking.

Fuel loading was significantly different in post-outbreak stands than in control stands, although results varied by study site. In general, 1- and 10-hr fuel inputs continue to accumulate up to 2 years after an outbreak, but slow by 8 years post-outbreak since most of the smaller branches have fallen to the forest floor by this time. Larger 100-hr fuels continue to accumulate on the forest floor up to 8 years after outbreak as large branches continue to fall from dead trees. After most branches have fallen, the snags begin to break off into sections or fall over whole, and CWD loads continue to accumulate up to 8 years after outbreak. Eight years post-outbreak does not appear to be long enough for CWD inputs and decomposition to reach a balance.

Areas containing heavy fuel loading can be mitigated by removing infected and buffer trees for lumber purposes. Salvage logging will not only decrease potential fire behavior, but will also provide timber revenue. Fungus-sniffing detector dogs have been successfully trained to find infected trees, which can then be removed and sold before the tree actually dies. By removing trees before they are dead or fall to the forest floor, heavy fuel loading can be avoided in these areas. If trees have already begun to fall or if salvage logging is not feasible, downed woody fuels can be removed at any point between 2 to 8 years after outbreak to decrease potential extreme fire behavior. If left untreated, species composition and fuel dynamics shift within SPB-killed stands, which may not only influence wildland fire behavior, but also timber production, water production, and wildlife habitat.

Ideally, forests should be managed to prevent SPB from reaching outbreak levels. Stand density is one of the most critical factors in spot initiation and expansion during an outbreak. Thinning is the preferred forest management strategy used to attain desired stand density, and it is generally recognized that pine stands with a basal area greater than 120 ft^2/acre should be reduced to less than 80 ft^2/acre to limit SPB outbreak (Clarke and Nowak 2009). However, when this is not a viable option, or for stands that have already experienced an attack, land managers should be equipped with the knowledge to predict what types of fuel loading and subsequent vegetative succession these stands will face, and what types of fire behavior to expect in a wildland fire (both wildfire and prescribed fire) situation.

Stand structure changes simultaneously as woody debris drops to the forest floor. SPB outbreaks create gaps in the forest canopy that allow for increased sunlight and nutrient availability; however, loblolly pine recruitment may be inhibited by limited exposure of mineral soil, while the growth and establishment of more shade-tolerant species may increase under the canopy from surviving trees and/or new growth. This study found that stands which were previously loblolly-dominated became occupied by oaks and other low-quality hardwoods after an outbreak, with hardwood density increasing over time since outbreak. If left untreated, the trajectory of forest stand composition after a beetle outbreak could transform from pine-dominated to hardwood-dominated forests. Therefore, if management objectives include pine restoration in these areas, fuel reduction treatments will need to be used to remove competing vegetation.

Characterizing ROS and flame length of a wildland fire is important for fire management. This information is used to assist in determining suppression tactics during wildfire, and can also be used for evaluation of fire effects after a fire. In the planning stages of prescribed fire, this information is used to define the conditions under which a burn will be conducted. According to the fire behavior pocket card used by wildland firefighters (Appendix A-1) (http://fam.nwcg.gov/fam-web/pocketcards/table.htm; Deeming *et al.* 1977) as well as the table related to fire suppression activities outlined in Pyne *et al.* (1996) (Appendix A-2), direct attack by handcrews should not be used on fires with flame lengths greater than 4 feet and direct attack by any means should not be used on fires with flame lengths greater than 6 feet. The results of this study strongly suggest that fire behavior is influenced by high fuel loading resulting from SPB outbreak. Both ROS and flame length increased with time after outbreak. Flame lengths were predicted to be greater than 4 feet in height for a majority of the EPO and LPO stands, even under dry moisture conditions, indicating that direct attack will probably not be feasible when suppressing fires in most SPB-killed stands.

A comparison of ROS between standard fuel models (Scott and Burgan 2005) and custom fuel models as predicted by the modeling systems found that in general, standard fuel models were not good predictors of fire behavior within the beetle-killed fuel complex. When attempting to model fire behavior for SPB-killed stands, customized fuel models should be used to obtain the most accurate results, however, if no field data are available for customization, standard fuel models can be used to predict similar results. Land managers could potentially use standard fuel models 'TL8,' 'TL9,' or 'SB1' when using fire prediction modeling systems such as BehavePlus to predict ROS for loblolly pine-dominated stands unaffected by SPB under dry and very dry conditions within the Piedmont. Fuel model 'TL9' could potentially be used for modeling stands at 2 years post-outbreak under dry conditions, while fuel model 'TU4' could be used to predict ROS for stands at 2 years post-outbreak under very dry conditions or for stands at 8 years after outbreak under dry conditions. Fuel models 'TL9,' 'TU4,' and 'SH4' are the closest predictors of flame length under dry, very dry, and extremely dry conditions, respectively, for stands at either 2 or 8 years after outbreak. These standard fuel models are only valid under the moisture conditions specified above, and are not accurate under any other conditions. Fire behavior is anticipated to decrease under more moderate moisture conditions, in which case other standard fuel models such as fuel models 'SB1,' 'TL7,' or 'TL4' may become more applicable.

The results from this study are based on environmental conditions during the Piedmont fire season and fuelbed data specific to these stands. While fire behavior modeling systems are invaluable tools, predictions vary between stands and during different times of the year depending on the input data. Actual fire behavior on a site will depend on a variety of factors including fuels (amount, arrangement, chemistry, moisture content), weather (wind, relative humidity, solar radiation, temperature), and topography (slope, aspect). Each of these factors should be taken into consideration when conducting fire behavior modeling for use in fire management. The time since beetle outbreak is also an important factor to consider in the relationship between outbreak and fire behavior since fuels change over time.

Quantitative fire behavior data collected from wildland fires occurring within SPB-killed stands are necessary to validate the results of the modeling predicted in this study. While more data are necessary to fully understand the changes that take place over time after a SPB outbreak, the results of this study nevertheless fill gaps in knowledge regarding fuel loading, post-outbreak succession, and expected fire behavior within SPB-killed stands in the Piedmont. The information provided within this study will allow land managers to make more informed decisions regarding strategic planning of fuels reduction projects, prescribed burning techniques, fire suppression, and other management objectives within these SPB-killed fuel complexes.

APPENDIX

Burning Index/Fire Behavior Cross Reference (top) and Flame Length and Fireline Intensity Related to Fire Suppression Activities (bottom)

BI	Flame Length	Fireline Intensity (BTU/ft/sec)	Narrative Comments
0-30	0-3	0-55	Most prescribed burns are conducted in this range.
30-40	3-4	55-110	Generally represent the limit of control for direct attack methods.
40-60	4-6	110-280	Machine methods usually necessary or indirect attack should be used.
60-80	6-8	280-520	The prospects for direct control by any means are poor above this intensity.
80-90	8-9	520-670	The heat load on people within 30 feet of the fire is dangerous.

Flame Length	Fireline Intensity (BTU/ft/sec)	Fire Suppression Interpretation
<4	<100	Fire can generally be attacked at the head or flanks by persons using handtools. Hand line should hold the fire.
4-8	100-500	Fires are too intense for direct attack on the head by persons using handtools. Handline cannot be relied on to hold fire. Equipment such as plows, dozers, pumpers, and retardant aircraft can be effective.
8-11	500-1,000	Fires may present serious control problems—torching, crowning, and spotting. Control efforts at the fire head will probably be ineffective.
>11	>1,000	Crowning, spotting, and major fire runs are probable. Control efforts at the head of the fire are ineffective.

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